

**MEETING MANAGEMENT GOALS IN AN URBAN FOREST:
VEGETATION DYNAMICS AND PRESCRIPTIONS IN STANLEY
PARK**

by

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Abstract

As urban areas grow, urban trees attract more attention from city governments, foresters, arborists and citizens because of the ecological services and amenities they provide and also the hazards they pose. The term 'urban forest' has a wide variety of meanings. These meanings are rarely functional. Urban trees live in habitats where human interventions in ecological processes and environmental conditions range from severe to minimal. One end of this gradient represents an 'urban treescape' while at the other end are 'self-maintaining forests'. These terms are defined within a conceptual framework that better describes the functional range of forests found in urban areas. Management of urban forests requires techniques and concepts from both forestry and arboriculture. Stanley Park is an example of a semi-self-maintaining forest. Aerial photograph analysis shows that the forest composition and structure of the Stanley Park forest has changed substantially since the 1930's. Windstorms in 1934, 1962 and 2006 have played an important role in this change. To characterize current conditions, data were collected from 208 sample plots to describe the forest vegetation. From these data and historical vegetation data, 15 vegetation types were defined. Data from representative plots were used to model different management scenarios in stands disturbed in 2006 and stands replanted following the windstorm in 1962. In the stands initiated following the 1962 storm, thinning over the next few years would reduce the overstory height to diameter ratios (HDR) but, because of the height of the stand, the windthrow hazard will remain high. Planting densities in the stands disturbed in 2006 were low, but there is considerable natural regeneration. Early thinning treatments will be required to achieve compositional and structural targets outlined in the Stanley Park Forest Management Plan (Vancouver Park Board 2009). Thinning should be done in the next 10 to 15 years in these stands to achieve the management goals most efficiently. The modeling tools used here can be

used in other urban forest settings. They can be particularly valuable when managers must plan silvicultural treatments to reach management goals and when presenting treatment options to the public and budgetary authorities.

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To Queen and my parents

1. Introduction

Vancouverites are passionate about Stanley Park. The park receives 8 million visitors per year (Vancouver Park Board 2007). When a storm in December 2006 blocked roads and trails and damaged sections of the popular seawall path, Vancouverites, corporations and governments showed their concern for the park by donating \$9 million (Vancouver Park Board 2007) in financial and in-kind contributions for park restoration. Managers of the park, located in Vancouver, British Columbia (BC), Canada, set about repairing the damage to the park and engaging citizens on the future management direction of the 256 hectares (ha) of remnant coastal forest (Fig. 1-1).

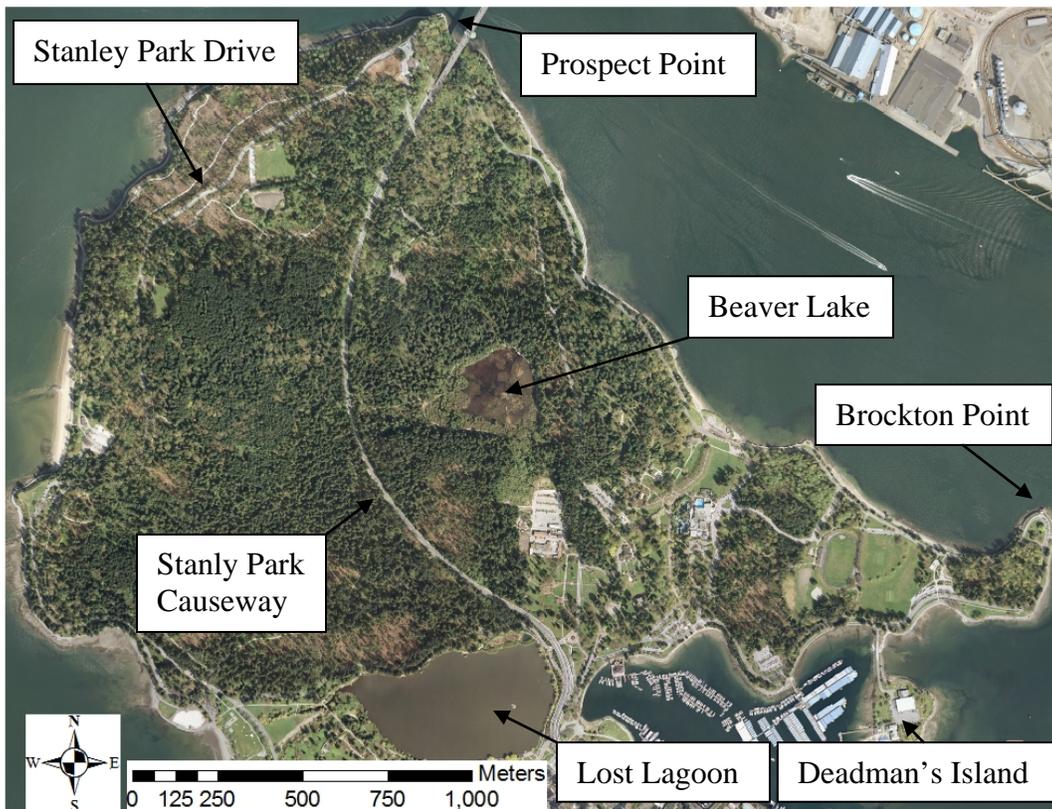


Figure 1-1. Aerial photo of Stanley Park from 2008 with points of interest labelled.

Remnant forest patches, such as the Stanley Park forest, are just one kind of urban treed environment. Trees are common in cities and cover a significant portion of the urban land base. Nowak et al. (1996) shows that in temperate regions most cities have a tree canopy cover exceeding 10%. One city in his survey had a tree canopy cover of greater than 50%. Rowntree (1984) suggests that areas with tree canopy cover greater than 10% can be considered forests. Because tree canopy cover in urban areas often exceeds this 10% threshold, urban trees have come to be thought of as located in 'urban forests'. However, urban trees grow in environments that can differ greatly from remote forests and, most importantly, in places where forest processes are affected by human activities. Even within a single city, trees occupy areas with many different types of management and land use regimes. A short list of habitats would include street-sides, residential yards, parks, and commercial and industrial grounds. Different land use and land management regimes can greatly affect the growth of individual trees and the structure and composition of stands by changing the pathways of ecological processes. A gradient can be used to describe the level of impact that human activities have on these processes. At one end of the gradient, where ecological processes are most impacted, are human designed and built areas, such as plazas, that contain trees in planters. These areas will be defined later as 'urban treescapes'. At the other end of the gradient are forests in rural areas at the edge of the wildland urban interface. Ecological processes in these forests are only moderately affected by human intervention as compared to remote forests or so called 'natural forests'. Humans may have indirect effects on all forests, for example through climate change (Biosvenue and Running 2006) or changes in atmospheric nutrient deposition (Fenn et al. 2003).

Stanley Park is an example of a large urban forest in which forest processes have been moderately modified by humans. Modifications to forest processes have been intentional, such as the planting of selected species following disturbance, and unintentional, such as the modification of the nutrient cycling pathways due to the removal of woody debris and addition of nutrients from human sources. Following the wind storm of 2006 managers prepared a new forest management plan for the park that takes into account ecological and visitor safety goals (Vancouver Park Board 2009). The Stanley Park Forest Management Plan acknowledges that some natural processes are incompatible with human use of the park (ibid.). The plan also lays out the vision for forest management at Stanley Park. The vision is:

“That Stanley Park’s forest be a resilient coastal forest with a diversity of native tree and other species and habitats that allows park visitors to experience nature in the city” (Vancouver Park Board 2009).

In this thesis I will first explore the concept of urban forest and suggest a new conceptual framework for thinking about forest processes and management in urban forests. I will then focus on Stanley Park to illustrate some key challenges and approaches for managing urban forests. I will give particular attention to the effects of recurrent windstorms on the park and strategies to improve resiliency of the forest to future disturbances.

1.1 Objectives

The objectives of this thesis are as follows:

1. To critique existing terminology used to describe urban forests;
2. To use the idea of a gradient of human impact on ecological processes to better define urban forests;

3. To explore the past and present stand conditions in Stanley Park, and how these conditions have been affected by disturbance and management; and
4. To explore the possible future stand conditions at Stanley Park, and how these conditions could be modified by management.

1.2 Approach

Information from scientific literature, historical accounts, aerial photographs, field plots, and stand growth modelling was used to accomplish the objectives listed above. A thorough review of urban forestry literature was the basis for the critique on urban forestry terminology. Literature on ecological processes provides the foundation for the idea of ‘urban treescapes’. Historical accounts, including primary documents and aerial photographs, were used to track changes in stand conditions to 2008. Field data collected in 2008 and 2009 as well as an existing, but outdated, forest cover map, and contemporary aerial photographs were used to re-map the vegetation in Stanley Park. This information was compiled in a geographic information system (GIS). The field data were also analyzed to produce descriptive statistics using SAS statistical software (SAS Institute 2003). Possible future conditions of the forest in Stanley Park under different management scenarios were modeling using the Pacific Northwest Coast variant of the Forest Vegetation Simulator growth and yield model (FVS; Forest Vegetation Simulator Staff 2008). This analysis of current and future conditions led to a set of recommendations for current and future Stanley Park managers, and a set of general recommendations for managers of urban forests and the scientists who study them.

1.3 Research questions

The research questions explored in this thesis are:

1. What is the current state of urban forest terminology?
2. How are forest processes incorporated into the definition of urban forest?
3. How has the structure and composition of the forest in Stanley Park changed over time?
4. What is the current composition and structure of the forest in Stanley Park?
5. What is the composition of regeneration (both planted and natural) in 2006 windthrow areas?
6. How will forest stands in the major vegetation types develop over the next 100 years?
 - a. What will be their composition and structure?
 - b. What will be their growth trajectory, with and without thinning, and how will this affect the windfirmness of the individual trees/stands?
7. What interventions are necessary to achieve target stand conditions?

1.4 Thesis outline

Chapter 2 of this thesis provides a literature review of urban forest terminology. This chapter is drafted for a journal submission. It sets the stage for discussion of urban forest issues including identifying amenities and ecological services and problems with standard urban forest definitions. New terminology is introduced to describe the gradient from minimal to great human intervention in natural processes. Six key natural processes and two environmental factors that are important in determining forest structure, which has a role in providing ecological services or amenities, are described. The implications for management in both the planning and intervention stages are provided.

Following Chapter 2 the thesis turns to the main focus of this work, Stanley Park. In Chapters 3, 4 and 5 focus on the disturbance history, vegetation dynamics and potential management interventions in Stanley Park.

Chapter 3 explores on the past and present structure and composition of the forest at Stanley Park. The historical development of the forest and the role of natural disturbance and human intervention are discussed. The results from analysis of aerial photographs and field data are presented, the current stand types are described and implications for management discussed. The results from this work are used as input for modelling of future stand trajectories, which is the subject of Chapters 4 and 5.

In Chapter 4, modelling of stands that developed following a windstorm in 1962 (also called, variously, Freda, Hurricane Freda, Typhoon Freda, and the Columbus Day Storm) is presented. This chapter includes the results of modelling trials used to predict the forest structure under different management scenarios. Recommendations to managers for achieving the target stand conditions described in the Stanley Park Forest Management Plan (Vancouver Park Board 2009) are presented.

In Chapter 5, the results from modelling stands disturbed in 2006 under different management scenarios are presented. As in Chapter 4, recommendations for management for achieving the target stand conditions described in the Stanley Park Forest Management Plan are presented.

Chapter 6, the final chapter in this thesis, discusses the implications of the preceding chapters for management of urban forests and urban treescapes, and recommendations for better managing these spaces. While the recommendations provided here stem from the lessons learnt from Stanley Park, it is intended that they can be applicable to a wide range of urban tree environments.

2. Forests and treescapes: using gradient of natural processes to define urban forest types

2.1 Introduction

Trees and treed spaces in urban areas provide economic, ecological and amenity benefits. Some ecological services and amenities have been shown to have economic value including local climate moderation (McPherson et al. 1997; Brack 2002), the removal of air pollutants (Nowak and Dwyer 2007), carbon sequestration (Brack 2002; Nowak et al. 2002) and recreation (Tyrväinen and Meittinen 2000). Other services include stormwater retention (Sanders 1986) and noise reduction (Boland and Hunhammer 1999), but the economic value of these services has not yet been quantified. The benefits provided by urban forests are likely to increase in value as the climate warms and urban populations increase. Gill et al. (2007) note that because vegetation has the ability moderate summer temperatures, increasing greenspace, such as by increasing tree cover, could be a method to mitigate the effects of climate change in urban areas. Akbari and Konopacki (2003) note that trees and vegetation directly reduce the urban heat island effect by shading buildings and indirectly lower temperatures through evapotranspiration.

The United Nations has projected that the world's urban population will increase to 70% of the total population by 2050 up from 50% today (United Nations Secretariat 2009). The increasing urban population will create pressure for urban infill development that, along with budgetary and liability constraints for urban governments, may lead to reduced urban canopy cover and reduced tree size in new plantings (e.g. McPherson 2000; Pauleit et al. 2005). Urban space is limited. Where trees are deemed to have little or no net value they may be replaced by

alternative land uses. Facing a decline in urban tree canopy cover and the associated loss of ecological services, some cities have developed city-wide management plans to protect existing trees and promote tree establishment (e.g. Portland Parks and Recreation 2004; City of Seattle 2007).

Well designed management plans should enable urban governments to manage trees in a variety of settings for maximum benefit. Trees in urban, suburban and rural settings grow in ecosystems with varying degrees of modification of natural processes by human activities. Managing urban trees and forested areas with a high density of humans or human-built structures requires concepts and skills from both arboriculture and forestry. There are areas of knowledge overlap between these two disciplines and also knowledge gaps. For example, both foresters and arborists manage tree health via interventions. However, foresters focus on forest community and landscape-level planning and interventions, and increasingly recognize the importance of retaining unhealthy and dead trees in ecologically healthy stands (e.g. Raffa et al. 2009). Arborists have traditionally focused on tree-scale issues, identifying suitable species for planting in urban and suburban settings, promoting the health of individual trees and removing dead and diseased trees. It would improve dialogue between arborists and foresters, and assist municipal forest managers, fire protection officials and scientists, if there were a collective understanding of key concepts and more precise terminology. To be useful, these concepts and terms need to provide a basis for management decision making. In particular, we need to be clearer about what we mean by the term ‘urban forests’, and the properties and administrative implications of various kinds of urban forests.

This chapter reviews current definitions of urban forests. An ecological process based gradient of urban forest types, from self-maintaining forests to highly human-modified areas is presented. The ecological processes that make forests along this gradient distinctive are highlighted, and the implications for provision of ecosystem services and the administrative responses that should follow are considered. Finally, new terms to define forest types along this gradient are presented.

2.2 Definition of the urban forest

There are many definitions of the word 'forest'. Lund (2002) reviewed existing definitions and classified them into administrative, land use, or land cover categories. The administrative category includes entities such as the U.S. National Forests, which do not require the presence of trees on all portions of the landscape. Examples provided by Lund (2002) show that land use based category typically includes a reference to timber production. The land cover category considers tree height or canopy cover to identify forest areas. Land cover based definitions can apply to urban areas. Helms (2002) reported that the definition of 'forest' used by the Society of American Foresters (SAF) has changed and broadened over time. The most recent SAF definition described by Helms (2002) identifies a forest as a place, "characterized by more or less dense and extensive tree cover". 'Urban forest' is included on a list of special forest type, but is not defined.

Just as the word 'forest' has many definitions so too does 'urban'. McIntyre et al. (2001) reviewed the definition of the word 'urban' in 63 papers from ecological journals which focused on urban ecology. They found that the definition varied by study and that the definition was

typically vague. They looked beyond the field of ecology for suitable definitions of 'urban' and found that many definitions are arbitrary or do not define minimum densities of people or structures.

Population density is one metric commonly used to define urban areas (e.g. McDonnell et al. 1997). Density thresholds are variable, however, and appear somewhat arbitrary. The current U.S. Census Bureau definition of 'urban' classifies urban areas as places with at least 386 people per km² or areas surrounding a place with more than 386 people per km² and with a density of greater than 193 people per km² (U.S. Census Bureau 2008). Statistics Canada defines urban areas as places with a minimum of 1000 people and at least 400 people per km² (Matier 2008). While population density is an important feature of urban spaces, it is so variable that it makes the generalization of urban conditions difficult. In 1990, for example, the population densities of the 20 largest cities in the U.S. varied from 9150 people per km² to 309 people per km² (U.S. Census Bureau 1998). Some of this variation is from fact that city boundaries are not located at natural breaks in population and may not be adjusted to compensate for urban growth. Furthermore, development trajectories followed by cities vary greatly throughout the world. In countries that have development patterns different from the Canada and the United States, the human density thresholds used to define urban areas in North America may be inappropriate.

The existing definitions of 'urban forest' are similarly inconsistent, and range from descriptive to administrative, but are rarely functional. These definitions hinge on the density of trees required to consider an area forested, and the density of people required to consider an area urban. Rowntree (1984) presented a widely used definition of 'urban forest'. Central to this

definition is the idea that any place with greater than 10% tree canopy cover can be considered a forest. Nowak (1994) reports urban tree canopy cover by land use and shows that in many American cities only commercial and industrial lands have less than this 10% threshold. Nowak et al. (1996) extends the analysis to other cities in temperate regions and shows that most cities have a tree canopy cover exceeding 10%.

Other definitions, which are descriptive rather than quantitative, exist. Konijnendijk (2003) lists many different definitions provided by national experts from several European nations. Konijnendijk et al. (2006) note that the concept of 'urban forest' is split into two senses in many non-English speaking countries in Europe. The narrow sense refers to city owned woodlands which were traditionally held by European cities. The broad sense includes street and residential trees and is similar to the concept of urban forest applied in Canada and the United States. Some definitions of 'urban forest,' such as that given in the Canadian Urban Forest Strategy, are so broad as to include non-treed greenspaces (Canadian Urban Forest Network 2005). Dwyer et al. (2000), in a nationwide survey of U.S. urban forests, use several definitions of 'city' or 'urban' and therefore of 'urban forest'. The most restrictive identifies urban forests as located only in Census Bureau designated places with > 50,000 people. The broadest definition of 'urban forest' included all trees in counties which contain cities with populations > 50,000.

Forests often grow to the edge of and into urbanized areas. In North America this zone is called the wildland-urban interface (WUI). Researchers studying the WUI use terms entirely different from those used by researchers studying core urban areas. U.S. federal guidelines

describe three types of WUI and focus on wildlands in terms of forest fire fuel hazard rather than vegetation community type (U.S. Department of Agriculture and U.S. Department of the Interior 2001). The first type of WUI is the interface where structures abut areas with wildland fuels. Intermix, where the boundary between fuels and structures are not clearly demarcated, is the second type. The third type is occluded communities, which are islands of wildland fuels surrounded by structures.

2.3 Problems with definitions of urban forests

The professionals who design and manage the built and natural features of urban and suburban areas make decisions that impact and are in turn impacted by the life and death of individual trees, and the ecological processes that occur in communities of trees. Urban forest types should therefore be distinguished where functional differences exist and where these differences will affect the nature and success of plans and prescriptions. Definitions based on the density of humans or trees are appealing from an inventory perspective because they are easily defined and quantifiable. For managers, however, they are arbitrary and ultimately unhelpful. The same problem applies to definitions based on administrative categories.

Forests are diverse. Even within a single country a wide range of forest types are present. Important structural features of one forest type may not be present in other types. Even so, several general ecological processes take place in all forests and are essential for continuation of forest cover in the long term. These include regeneration, competition for growing space, and nutrient cycling, among others. These processes are in turn mediated by disturbance regimes. In some cases these processes are highly modified by human activities. Using the degree of

departure from natural forest conditions and processes due to human activities provides a functional basis for describing the effects of human activities on forest functions. Characterizing these conditions and processes provides insights into the potential impacts of forest functions on human lives and structures. Aside from people, important features of urban areas include buildings, landscaped areas, drainage, transportation infrastructure and utilities. Persistent human-built structures and periodic disturbance of vegetation, hydrology and soils may disrupt natural processes even when there is no continuous human presence. For example, roads fragment natural forest areas, change local hydrology and fire patterns, and interrupt animal movement (Forman and Alexander 1998). The release of chemicals by vehicles and road surfaces can have a lasting effect on forest processes (e.g. Pouyat et al. 1995). In characterizing the effect of human-induced disturbance on forest processes, it is important to separate the effects on key ecological processes and to identify dose-response relationships. The points at which key functions are fully impaired can be useful benchmarks for identifying transitions along the gradient from semi-natural to fully built environments. In fully built environments, key ecosystem services provided by natural systems will need to be replaced by human-designed alternatives. Using functional definitions will make this more explicit and promote the proper valuation of ecosystem services.

2.4 Forests and treescapes

The definitions reviewed above do not take into account ecological processes when describing urban forests. Ecological processes in turn lead to the development of structural features such as vegetation distribution and large woody debris, among others. Changes to these processes through design, management or surrounding land use can change the attributes of

forested areas and the resultant services and amenities they provide. Given the importance of these processes to forest formation and sustainability, and the consequences of these processes for the human inhabitants of forested or semi-forested areas, it makes sense to refine our terminology.

Forests can be categorized across the gradient from natural to built landscapes based on the degree of impact in natural processes of to human activities. To describe the extreme urban end of the gradient, the term ‘urban treescapes’ is proposed. This term underscores the fact that, while tree cover is present, many fundamental forest processes are highly or fully impaired by recurring human activity. In contrast, in ‘self-maintaining forests’ these processes and the features that develop are governed by interactions between local climate, terrain, soil, and the biotic community with no or very minor influence from human activity, such as from fire suppression in adjacent forest areas. ‘Self-maintaining forest’ and ‘urban treescape’ represent opposite ends of the spectrum in terms of intervention by humans in natural forest processes. In urban treescapes these processes are governed by humans, with local climate and soil conditions only broadly limiting the range of possible outcomes (e.g. frost intolerant trees planted in cold climates will fail, but irrigation allows for drought intolerant trees in dry climates). Urban treescapes may be strictly designed, such as urban plazas and boulevard plantings, or largely unplanned, such as remnant natural trees left along property lines. All, however, are subject to continued human intervention in fundamental processes to maintain their intended character, and this may require significant inputs of labour and capital.

A broad range of forest conditions fall between self-maintaining forests and urban treescapes. The term 'semi-self-maintaining forest' is proposed to describe forests with intermediate levels of human disruption of natural processes. Managed forests of all types fit into this category, making this the most common type of forest in human dominated landscapes. Only inaccessible or wilderness areas, where management interventions are not conducted and natural disturbance processes continue with minimal human disruption are considered self-maintaining in this conceptual framework.

In order to highlight the distinctions between self-maintaining forests and urban treescapes, this chapter focuses on 6 key ecological processes in forests: tree regeneration, understory vegetation development, competition for growing space, tree death, decay and nutrient cycling, and disturbance. These processes were chosen because they are important to the development and maintenance of forest structural attributes and because they have important implications for the human inhabitants of forested areas. Oliver and Larson (1996) describe how regeneration, disturbance, and competition for growing space are core processes in forest stand dynamics. Kimmins (1997) considers tree death, and decay and nutrient cycling, key processes in forest ecology. Understory vegetation development is proposed here as also being an important process to consider because this process is routinely modified in human dominated landscapes. In addition, 2 environmental conditions are important to forest development, the hydrologic regime and the temperature regime (Kimmins 1997).

2.4.1 Tree regeneration

In self-maintaining forests, regeneration depends upon having a suitable seed or propagule source, having available seedbed and having environmental conditions that favour establishment. The disturbance regime opens new sites for establishment, determines which regeneration mechanisms are favoured (Oliver and Larson 1996) and drives stand composition (Oliver 1981). Ultimately, successful species are those which are adapted to the local climate (Rochefort and Peterson 1996), soils (Beckage and Clark 2003), pathogens (Packer and Clay 2000), insects (Kulman 1971), and disturbance regime, including grazing, (Tilghman 1989) and are able to compete with associated understory and overstory plants (Lutz and Halpern 2006).

Regeneration in urban treescapes is a highly human modified process. Many exotic species have been introduced into urban areas for aesthetic reasons and because of their suitability for particular planting situations (McBride and Jacobs 1976; Jim and Liu 2001). Management and land use history, climate and soil conditions, and societal preferences interact to determine species composition in areas where trees are planted (Jim and Liu 2001). Considerations such as size at maturity, growth form, aesthetics, and management costs may be just as important as climatic tolerances when urban tree species are selected (Sæbø et al. 2003). These trees are often planted as saplings, thus avoiding the challenges faced by regenerating trees on site. Significant and ongoing human intervention for provision of moisture and nutrients, along with suppression of competition and pests may be necessary to establish and maintain these trees in particular urban settings (Harris et al. 2003).

Human activities can unintentionally affect regeneration in adjacent stands. Bagnall (1979) noted that the increased edge area created by urban development can change species composition in remnant forest patches. In patches where natural regeneration was allowed, Zipperer (2003) found that regenerated forests that had previously been non-forested had a different species composition and more exotic species than forest patches that had always been forest. He suggested that late successional species are less able to reproduce in patches that were previously non-forested due to increased disturbance rates and trampling. In addition regeneration of fire dependent species was low in all forest patches where fires were suppressed (Zipperer 2003).

2.4.2 Understory vegetation development

Understory vegetation is highly modified in urban treescapes. Hard surfaces and turf grasses are common ground cover types in urban areas. McPherson et al. (1994) found that the most common ground covers in an urban and suburban area were maintained grass, asphalt, herbaceous cover and buildings, which accounted for 67 % of the total ground cover. Where vegetation is present, non-native species are commonly found in treescapes due to intentional planting or because they reproduce in places where human disturbance is more common (Kowarik 1995).

Vegetation composition is determined by factors including climate and soil conditions, disturbance regime and land use history among others. Godefroid and Koedam (2003) found that native woodland species richness increased with increasing urban forest patch size. Small patches were drier and had less acidic soil with more nutrients than larger patches. Species

composition in patches differed by soil acidity and moisture regime. In forests naturally regenerating after agricultural use, seed source and site quality have been found to be important in determining vegetation composition (Honnay et al. 1999). Exotic species may spread into vegetated areas adjacent to human disturbed landscapes, however, some vegetation communities might be more resistant to invasion than others. Gunterspergen and Levenson (1997), for example, found little change in the composition of understory species in remnant forest patches along an urban-rural gradient in the Milwaukee, Wisconsin, region. In semi-self-maintaining forests the understory may develop as in self-maintaining forests, but vegetation management and accelerated crown closure may modify the cover, composition and longevity of understory vegetation communities.

2.4.3 Competition for growing space

In forests, trees compete for growing space as they increase in size. In semi-self-maintaining forests, fragmentation may lead to a larger proportion of trees being subject to edge effects than in large contiguous blocks of forest. These edge effects may change species composition since edge conditions favour particular species (Bagnall 1978; Chen et al. 1992).

Early competition with herbaceous plants (Davis et al. 1998) and shrubs (Pabst and Spies 1999) can affect the survival of tree seedlings in forests. Competition between growing trees leads to canopy stratification and competitive exclusion (Oliver and Larson 1996). In urban treescapes, the spacing at the time of planting is typically designed to enable all trees to survive and develop full crowns as they grow to maturity. Contact between the crowns of mature trees is common since an overarching canopy is often desirable. It is rare however, that tree densities

would be high enough to lead to competition-related mortality, and weak trees are typically removed before senescence.

2.4.4 Tree death

Mortality processes differ between self-maintaining forests and treescapes. Death results from density-dependent and density independent processes. Density dependent mortality occurs when trees lose the competition for growing space. Density independent tree death in self-maintaining forests has been described as a spiralling process whereby disturbance agents and physical damage make trees more susceptible to other disturbance agents, eventually leading to death (Franklin et al. 1987). There is a fundamental difference between tree health and forest health, particularly when forest health is viewed through the lens of ecosystem management. Raffa et al. (2009) note that 'forest health' describes the inherent processes in an ecosystem and ecosystem resiliency. Through this lens, healthy forests include diseased, injured and unvigorous individual trees (Raffa et al. 2009). In urban treescapes, where risk to human safety and structures are a prime concern, dead and diseased trees are considered unsightly and hazardous and are generally removed following an assessment. The International Society of Arboriculture notes that broken branches, open trunk or branch cavities, branches arising from a single point on the trunk, and recent changes in grade or soil level all can create a hazardous tree (International Society of Arboriculture 2007). Healthy trees may be removed if they become too large or if they impede use of the site for higher priority purposes. Unplanned tree removal also occurs in urban treescapes due to vandalism, development or pollution (e.g. Nowak et al. 1990), and urban tree life spans are often much shorter than expected for a particular species (e.g. Moll 1989 in Trowbridge and Bassuk 2004).

2.4.5 Decay and nutrient cycling

Much of the litter and debris from treescapes is collected and decays offsite. This contrasts with natural forest areas where accumulated litter and debris loads can be great. For example, Turner (1981) found nearly 100 metric tons of biomass per ha in a 73 year old Douglas-fir (*Pseudotsuga menziesii*) stand, most of which was coarse woody debris. Semi-self-maintained forests may have coarse woody debris, but it may be found in lower quantities than self-maintaining forest because of timber extraction or fuel management (Spies et al. 1988). In other semi-self-maintained forests long term fire suppression can increase coarse woody debris loads compared to self-maintaining forests with recurrent surface fire regimes (Brown et al. 2003). Semi-self-maintaining forests can have soil development, soil biota and nutrient cycling similar to self-maintaining forests, but modification in deposition rates through vegetation management, periodic removal of debris, harvesting or deposition of chemicals may modify decomposition and soil forming processes.

Urban treescapes typically experience high levels of nutrient inputs not found in self-maintaining forests. These include garden fertilizers, pet waste, and leachates from septic systems (Baker et al. 2001). Deposition of airborne pollutants containing nitrogen is an additional input in urban treescapes and semi-self maintaining forests surrounding urban areas (Fenn et al. 2003). Driscoll et al. (2003) note that nitrogen can be carried over long distances and deposition is not limited to urban and peri-urban areas. Nevertheless, Lovett et al. (2000) note that urban air has greater concentrations of nitrogen oxides and alkaline dust than air in rural areas. Kaye et al. (2006) suggest that nutrient cycling models used in natural areas, termed here

as self-maintaining forests, are inappropriate for use in urban areas because of the human caused changes to hydrology, atmospheric chemistry, climate, and nutrient additions, among others.

Levels of pollutants high enough to reduce soil quality or inhibit tree growth have been found in urban soils (e.g. Wade 1983). In some cases, urban soils have low permeability or wetability due to mechanical compaction or chemical alteration (White and McDonnell 1988). Gregg et al. (2003) found that ozone pollution was sufficient to decrease the growth rate of seedlings in forests surrounding New York City. Differences in soil fauna also affect nutrient cycling in urban areas, in some cases increasing the decomposition rate of leaf litter (e.g. Pouyat 1995).

2.4.6 Disturbance

Disturbance is an important factor in determining the composition and pattern of vegetation development (White 1979). Forest disturbance agents such as fire, snow, ice and wind can also impact human infrastructure and safety. Insects and pathogens can affect amenity values and lead to tree hazards. Urban development changes patterns of disturbance. For example, Syphard et al. (2007) found that fire frequency increased with increasing population density up to about 40 people per km² then decreased as population density continued to increase. Large fires in the urban-wildland interface have caused damage to thousands of homes and the loss of human life (Cortner and Gale 1990). In urban treescapes where maintained grass and paved surfaces are significant ground covers, fires may be rare and limited only one or a few individual trees. Fires are also generally small in this setting because suppression is prompt. However, even small fires can be costly due to the proximity of vegetation to structures. Utility

and transport networks may also be temporarily disrupted if a fire starts. Smoke from vegetation fires reduces air quality and affects human health (Mott et al. 2002).

Snow and ice deposited on trees in winter storms can result in limb breakage and tree failure in all categories of forests, from self-maintaining forests to urban treescapes. Tree species vary in their ability to support the additional loads brought by these storms due to differences in wood strength and crown form (Hauer et al. 1993). Lemon (1961) notes that, in the Eastern United States, ice storms can remove pioneer tree species and release suppressed shade tolerant species. Exotic trees that are not adapted to local snow or ice loading can be severely damaged (Ireland 2000). Trees grown in overly dense stands are more likely to experience stem bending, stem break or uprooting (Peet and Christensen 1980). Broken limbs can cause damage to property and utilities, and cause injury when they fall in densely populated urban treescapes. Additionally, amenities, such as shade and aesthetic values, can be lost with the reduction in tree canopy or loss of form.

Windstorms also damage trees in all categories of forests, damage property, disrupt communications and cause injury or death (Schmidlin 2008). In many regions, severely damaging wind storms routinely recur on an interval of several decades. Sources of strong winds vary regionally, but can be broadly grouped into extra-tropical cyclones (mid-latitude oceanic low pressure systems), tropical cyclones (tropical low pressure systems) and convective storms (thunderstorms and tornadoes). The severity and extent of damage depends on storm duration and wind speed, length and width of the storm track, and the effect of local terrain (Canham et al. 2001; Mitchell et al. 2001). Different tree species have different abilities to survive wind

damage (Canham et al. 2001). In self-maintaining forests wind damage can open growing space and create sites for seed germination. In urban treescapes the consequences depend on the nature of the target. Damaged trees are more susceptible to insect attack or infection by pathogens and are often removed.

Grazing can have impacts on forest vegetation composition. Changes in vegetation composition have also been noted where human activity has modified grazing regimes (Hobbs and Huenneke 1992) or affected the balance of predators and grazers (Beschta and Ripple 2008). In urban areas, mowing, pruning and debris collection can have a similar effect on regeneration and vegetation development, favouring certain species over others by eliminating tree and shrub regeneration and favouring grass and other low-growing, short lived perennials and annual herbs. Grazing can also be a factor in changing vegetation composition where deer populations have increased following removal of predators in and around cities (Etter et al. 2002).

2.4.7 Hydrologic regime

Hydrological modifications vary by land use in response to coverage by impermeable surfaces (Korhnak and Vince 2005). Korhnak and Vince (2005) note that the removal of forest vegetation and the construction of impervious surfaces increase the speed at which precipitation reaches stream channels. This causes an increasing peak discharge and total flow of a watershed. In contrast, in self-maintaining forest areas water storage capacity may be high and runoff flow rates low (Korhnak and Vince 2005). Base flow from groundwater can also be reduced because of reduced infiltration associated with impervious surfaces and sewer construction (Simmons and Reynolds 1982). Paul and Meyer (2001) note that changes in stream channel morphology can

occur, leading to wider more deeply eroded stream channels in urban areas compared to forested regions. In semi-self-maintained forests, coverage of impermeable surfaces is low, but road systems, particularly on slopes, can intercept through-flow converting groundwater to surface water and increasing run-off (Forman and Alexander 1998).

Removal of surface water and lowering of water tables via installed drainage systems and the efficient removal of run-off from hard surfaces via storm drain systems further modify hydrology and reduce water availability in urban landscapes (Simmons and Reynolds 1982). However, some urban treescapes receive moisture inputs via irrigation during summer dry periods. Where changes occur in local irrigation policies, such as bans on sprinkler use, mortality of long-established trees in urban treescapes may result.

2.4.8 Temperature regime

Forest canopies moderate local climate conditions. Forests have lower daytime air and soil temperatures and higher humidity than adjacent unforested areas (Chen et al. 1995). Forest patch size is important. In the Douglas-fir forests studied by Chen et al. (1995), fully 'interior' conditions were not found until up to 240 m inside the forest edge from adjacent clearings, with even larger distances expected in windy conditions. Large forested patches are therefore required to realize full climate modification.

In urban areas, Oke (1989) showed that tree stands and individual trees can ameliorate the heat island effect. Trees affect their microclimate by creating shade and through evapotranspiration (Akbari and Konopacki 2003). They also reduce local wind speeds

(McPherson et al. 1997). Together these factors can translate into energy and financial savings in hot climates (Rosenfeld et al. 1998).

2.5 Amenities and ecological services

Amenities are features that increase the comfort of an area to humans. In the context of trees in urban areas, amenities include the moderation of temperature and wind speeds discussed above, along with enhancement of recreation (Tyrväinen and Miettinen 2000), sound dampening (Bolund and Hunhammer 1999), and visual breaks in the urban landscape.

Ecological services are those attributes of ecosystems which provide benefits to humans. Without these services, human designed features would have to be constructed to provide the same level of benefit. Costanza et al. (1997) note many examples of ecosystem services including waste treatment, erosion control, water regulation, and gas regulation, all of which apply across the spectrum from self-maintaining forests to urban treescapes to varying degrees. In arid landscapes where large trees are not part of the surrounding environment, urban treescapes are often established specifically to provide local cooling through shading and evapotranspiration (Oke 1989).

Early proponents of bringing trees into public spaces in the densely populated cities of the 19th century suggested that trees could help improve air quality where waste mouldered in the streets (Lawrence 1993). More recently, other ecological services provided by urban trees have been noted. These include sequestration of carbon and removal of pollution (Brack 2002), and

reduction in runoff after storm events (Sanders 1986). The value of ecosystem services provided by forests depends on the location and land use.

There are potential downsides to urban trees. Direct costs associated with urban trees include maintenance costs, and damage caused by falling branches and trunks. Depending on location, tree growth can also damage graded surfaces and interfere with above and below-ground utilities and septic systems. Risks from natural disturbances such as wind, drought, disease, insect attack, snow and ice, or fire and the impact of falling debris on roads, power lines and buildings must be factored into cost-benefit analyses. Gardiner and Quine (2000) provide an example of such an analysis. The likelihood and consequences of disturbance changes over time as trees grow in size and age. For example, a particular tree may be of high value when small but become a hazard as it grows to maturity or begins to decline. Indirect costs associated with urban trees include the opportunity cost of the space required for a tree to grow. Trees may also fail to grow in the manner intended by the architect which may cause target goals to not be met.

2.6 Implications for design and management

The functional definition of forests and treescapes proposed here can be of use by managers in planning and implementing management activities. Development and community plans for populated areas should account for the amenity and ecosystem service values of forests or tree cover, the costs associated with routine tree and forest management, and the risks and costs associated with periodic disturbance events.

2.6.1 Planning

Plans will be more realistic and more successful if managers consider which category of forest occur within their jurisdictions and explicitly address the implications of the 6 key forest processes and 2 environmental conditions discussed above. For example, if managers recognize that some of their urban forest areas are semi-self-maintaining and subject to a number of natural processes, they can consider which processes are compatible, and which processes are incompatible with the intended patterns of human use and infrastructure development. They can create land use zoning regulations to minimize future conflicts and costs. They can design forest management regimes that enhance positive attributes and minimize the potential for negative impacts. For example, in areas subject to periodic fire, they can design and implement fuel management and fire preparedness plans.

Where natural processes are impaired, human interventions may be necessary to maintain viable forests or treescapes. For example, where hardened ground surfaces or frequent mowing is common, planting and physical protection may be necessary to enable tree establishment. Where the aim is to conserve semi-self-maintained forests in a relatively natural state adjacent to built-up areas, significant interventions may be required to control invasive species and maintain or restore drainage patterns.

Within urban treescapes, the degree of design and intentional management will vary. A paved plaza with potted trees designed to complement architectural features is an extreme example of an urban treescape with highly intentional design and management. Other parts of the urban treescape, such as residential backyards, may be significantly less planned, but may

still be subject to the requirements of local zoning, insurance standards or interventions for the protection of above-ground utilities or viewscales. Intensive management such as weeding, mowing, irrigating and pruning is practiced in these areas, but often without collective or even particular goals in mind.

It is also important for city planners to consider external risks to urban treescapes and the transmission of pests between urban and self-maintaining forests. For example, urban treescapes and semi-self-maintaining forests may be close to port facilities from which invasive insects and diseases can spread. Infections in these forests may then spread to more remote forest areas. Conversely, urban treescapes may be at risk from native tree pests such as bark beetles or defoliators spreading from adjacent forested areas (e.g. City of Prince George 2007). Species planted in urban treescapes may become invasive or naturalize into surrounding forest regions (Richardson et al. 2000). In some cases, native species may be at risk of extinction because of these introductions (Davis 2003), so urban planners should work in conjunction with resource managers and conservation authorities to identify appropriate species for urban plantings.

2.6.2 Intervention

Puettmann et al. (2009) describe forests as complex systems with non-linear relationships between features. Future forest conditions can be difficult to predict, even following natural disturbances in self-maintaining forests. Slight changes in climate or soil conditions can change local forest composition. Consequently, even low levels of human modification to any of the 6 key processes and 2 environmental conditions outlined above can result in substantial changes in a forest community. Semi-self-maintaining forests and urban treescapes are subject to recurring

interventions, whether intentional or unintentional. The challenge for planners and managers is to design and implement intervention regimes (prescriptions) that direct the development of forests, treescapes and their component trees and associated vegetation and soils toward desired future conditions, while improving their resiliency and minimizing the potential for undesired outcomes.

Resiliency is a key goal in urban forest management. It is defined here as the ability of urban trees and stands to resist and recover from disturbance. While some disturbance is inevitable, resilient trees and forests are damaged less than other trees and forest at a given disturbance intensity. Resiliency can be created by modifying tree or forest structures and composition. For example, in forests, the height to diameter ratio and ladder fuels can be reduced with silvicultural treatments (See Chapters 4 and 5), which can reduce the effects of wind and fire, respectively. Open grown trees can be pruned to reduce the risk of wind. Trees with poor resistance to local disturbance regimes can also be replaced by more resistant species.

Silviculture and arboriculture are both plant sciences. Harris et al. (2003) defines arboriculture as the planting and care of trees, shrubs and vines, individually or in small groups. Silviculture, on the other hand, is the management of stands of forests (Smith et al. 1996). Silviculturists typically design stand-level prescriptions following a thorough analysis of the site and stand conditions and the relationship between natural processes and stand development. Since trees are long lived, forests often retain the signature of historic disturbance and recovery. This process requires knowledge of ecological principles and tree biology, along with good

observational skills and local experience. Arborists take a similar approach in predicting the future growth and probable outcomes for trees within urban treescapes.

Silviculture and arboriculture are disciplines grounded in empiricism where past experience informs future expectations. Both disciplines also require an understanding of the available options for intervention and the ability to evaluate the pros and cons of alternative intervention techniques. Wise practitioners in both disciplines monitor the outcomes of their interventions, and compare these outcomes with their original expectations in order to learn and refine their future practice. Even with the best practice, undesired outcomes will periodically occur. Naturally, avoiding outcomes that have lethal or highly expensive consequences takes the highest priority. In an urban forestry context, an acceptable range of outcomes should be established during the planning phase, and monitoring should be extended to the landscape scale over the life of the plan to ensure that overall outcomes fall within the desired range. Understanding the relationship between forest ecological processes, structural features, ecological services and amenity values, and adoption of terminology that clarifies distinctions in forest types will assist planners and managers in designing and implementing plans for populated forested landscapes that produce desired outcomes.

2.7 Conclusion

Standard definitions of ‘urban forest’ do not recognize differences in functionality of ecosystem processes in treed areas within and outside of forest areas. The term ‘urban treescape’ is more accurate in describing areas where human intervention substantially most modifies ecosystem processes and environmental conditions. The distinction between ‘self-maintaining

forests' and 'semi-self-maintaining forests' is the level of human intervention in forest processes and on environmental conditions in forest areas. The exact nature of how and to what extent the ecological processes and environmental conditions are modified depends on the setting of the forest. Distinguishing these three categories of tree-dominated vegetation is important as each category provides different benefits, presents different risks and requires different management responses. City planners and urban tree specialists can benefit from this distinction in categories of forests by creating management plans that reflect the features of these categories of forests and incorporate a better understanding the relationship between urban trees and the natural and built environment.

The next chapter will focus in on Stanley Park. Stanley Park is a good example of a semi-self-maintaining forest in the centre of a major city. The challenges faced by park managers in Stanley Park illustrate the difficulty of managing forests in this urban context, and provide an opportunity to demonstrate the application of forestry concepts and tools in urban forest management

3. Stanley Park vegetation dynamics

3.1 Introduction

When the strong winds of a winter storm blew down trees throughout Stanley Park in December 2006 it was easy to think that a magnificent forest was destroyed (CBC News 2006). The forest of Stanley Park, however, has been constantly changing since before the creation of the park in 1888. Both human and natural disturbances and vegetative growth have changed the appearance and composition of the forest on relatively short time scales compared to the age to some of the trees in the park.

This chapter first looks at the physical setting and disturbance processes in Stanley Park and then the history of management. Management has created a semi-self-maintaining forest in Stanley Park. How these factors have changed the structure and composition of the forest canopy is explored. Finally, the current structure and composition of the Stanley Park forest is presented and discussed.

3.2 Biophysical conditions

Stanley Park is located on a peninsula that projects from downtown Vancouver, BC, and that separates Burrard Inlet from English Bay. The park is underlain by glacial till except along the northwest coast from Prospect Point to Siwash Rock which is on sandstone bedrock (Geological Survey of Canada 2008). Bedrock of volcanic origin is found along the western shoreline of Stanley Park (ibid.). The sandstone layer has been eroded by the ocean forming shoreline cliffs up to 15 m high.

The main body of forest in the park, located west of Pipeline Road, generally slopes from south to north at less than 10% grade. The hill reaches a maximum height of about 70 m near Prospect Point. North from this high point the land slopes down to the ocean at a slope generally greater than 10%. To the east of the main body of forest is the Brockton Peninsula. Here the ground slope is slight and maximum elevation is less than 10 m above sea level.

Most of the park is in the Coastal Western Hemlock Dry Maritime Subzone (Green and Klinka 1994). A small strip of land at the top to the sandstone cliffs is in the Coastal Western Hemlock Very Dry Maritime Subzone. Common native tree species present in the park are western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco.) and western redcedar (*Thuja plicata* Donn.).

3.3 Stanley Park forest history

3.3.1 Natural disturbance

Natural disturbances are regular occurrences in Stanley Park. In coastal British Columbian forests, including Stanley Park, winter storms associated with Pacific low pressure systems (extra-tropical cyclones) frequently blow down trees. The damage of December 2006 was just the most recent example of the effect a powerful windstorm can have on the forest. It was not an anomalous event. Damage of similar extent as seen in 2006 occurred in 1962, 1934 and possibly 1901 (Kheraj 2007). Less damaging storms happen more frequently. Kheraj (2007) notes that nineteen storms were reported between 1900 and 1960, each of which blew down tens to hundreds of trees.

The long-term history of fire in Stanley Park is not well known, however there is information available on forests with similar compositions. Long et al. (1998) note that the fire return interval in the late Holocene (from 2750 years before present) is 230 years in forests coastal Oregon. First Nations are known to have used fire to managed vegetation in coastal forests (Lepofsky and Lertzman 2008) but it is not known to what extent these fires modified the forest vegetation in Stanley Park.

One large fire on the Stanley Park peninsula occurred in about 1885 and burned the area between Lost Lagoon and Beaver Lake (Vancouver Park Board 2007) but the exact extent of the fires is not known. A dense stand is visible south of Beaver lake in aerial photographs from 1949 (Fig. 2). This stand likely regenerated following the 1885 fire. In the summer of 1888, just prior to the official opening of the park, several small fires burned in the park (Kheraj 2007). These fires were started by road construction crews, and dry conditions that summer allowed them to spread.

Early park managers were concerned about fire damaging Stanley Park. Rangers were hired to patrol the park during fire season, and a system of fire hydrants was installed in the park beginning in 1910 (Kheraj 2008). Today, because of human activity in the park, fire ignition is more common than in similar forests with less use (Vancouver Park Board 2009). Conditions exist for ignition and spread of a major fire on an annual basis. However, fires are typically detected and extinguished quickly.

Insects have also played a role in Stanley Park. A report from 1914 noted that 25% of western hemlock trees were dead and 60% were affected by insects (Vancouver Park Board 1939). The report also noted that fewer than 8% of Sitka spruce (*Picea sitchensis* (Bong.) Carr.) trees were healthy. In 1914, the first trials of insecticides were initiated. The insecticide BtK was used as recently as 1993 (Paul Lawson, R.P.F., pers. comm.). The primary species targeted were hemlock looper (*Lambdina fuscicollaria*) and an unidentified species of gall aphid (Bakewell 1980, Kheraj 2007). This gall aphid may have been balsam woolly adelgid, (*Adelges piceae*) the only major adelgid species in the area that affects conifers (Stephen Mitchell, R.P.F, pers. comm.). Bakewell (1980) notes that balsam woolly adelgid damaged Sitka spruce, grand fir (*Abies grandis* (Dougl.) Forbes) and western hemlock in the 1960s.

Park managers have outlined management strategies for monitoring future outbreaks of native and invasive insects in the latest management plan (Vancouver Park Board 2009). If populations of pest insects reach predefined thresholds management action will be planned.

3.3.2 Human disturbance

The Stanley Park area has long been used by humans. In a social history of Stanley Park, Barman (2005) reports that First Nations people cut western redcedar trees to make canoes and other products. Selective logging by European settlers took place from 1860 to 1886 (Vancouver Park Board 2007). First Nations people had village sites on the peninsula. In an area near Lumberman's Arch, where the First Nations village of Whoi Whoi was located, road workers found a midden of clamshells and bones up to 2.5 m thick covering 1.6 ha, indicating a long history of habitation (Barman 2005). Barman also notes that in 1876, twelve years before the

opening of the park, the British Columbia Reserve Commission recorded 50 residents in the area of the future park. After the establishment of the park, a small community of families remained living along southern shore of the Stanley Park peninsula near Deadman's Island (Barman 2005). The last remaining permanent resident of Stanley Park, a descendent of one these families, died in 1957 at which point his residence was demolished (Barman 2005). Habitation of the park by homeless people continues. Well established unauthorized camping sites were seen in the park during forest inventory.

The Lions Gate Bridge was opened in 1938, joining Vancouver to the north shore of the Burrard Inlet via the Stanley Park Causeway, which bisects the park north to south. The effects of the causeway on park vegetation have not been quantified. Anecdotal evidence, however, suggests that exhaust from cars is stressing the surrounding trees (Bill Stephen, pers. comm.). It is also likely that the causeway changed the surface and subsurface hydrology of the park, which would affect the soil moisture regime in adjacent stands.

The park was founded to provide a site for recreation in the city of Vancouver. Following the opening of the park, initial work included landscaping the park entrance and creating recreation facilities. From 1888 to 1930, 100 ha were converted to recreation facilities including a rose garden, boathouse, and athletic fields. In 1956 the Vancouver Aquarium was opened. Bakewell (1980) reports that between 1933 and 1980, 25% of the forest area in Stanley Park was lost to the development of recreational facilities and maintained grass and gardens.

Today, Stanley Park hosts 8 million visitors per year (Vancouver Park Board 2007). The impact of these visitors on the forest environment has not been quantified. Possible impacts visitors have on the park include trampling vegetation, compacting soil, increasing fire ignition, habituating wildlife, and increasing nutrient input from pet and human waste. Visitor can also introduced or spread invasive species in the park.

Early park managers saw the forest as in need of improvement. They undertook programs to remove dead tree tops and diseased trees as well as red alder (*Alnus rubra* Bong.) trees (Kheraj 2008). This was done to improve the aesthetics of the forest and meet public expectations of a ‘virgin forest’. Trees have been planted in the forests of Stanley Park for decades. Forestry reports from Stanley Park throughout the 20th century regularly note a perceived excess of shrubs and deciduous trees and an over-abundance of western hemlock (Vancouver Park Board 1939; Bakewell 1980). In response to these concerns, a revegetation program was undertaken in 1948, and by the late 1950s trees were being planted on nearly 5 ha per year (Livingstone 1973).

Managers at Stanley Park responded to the windthrow caused by the storms in 1934, 1962 and 2006 by initiating restoration efforts. Following the 1934 and 1962 windstorms restoration entailed removing windthrown trees and planting Douglas-fir (Kheraj 2007). Following the 1962 windstorm, specifically, debris was burned and Douglas-fir planted at densities of up to 2500 per ha (Bill Stephen, pers. comm.).

The restoration work following the 2006 windstorm was different from prior restoration projects in that an effort was made to retain forest structural features in windthrown areas. Immediate clean-up efforts were initiated after the 2006 windstorm and two smaller subsequent storms to insure that park facilities could be accessed safely. Restoration work was guided by the Stanley Park Restoration Plan, which was published in April 2007 (Vancouver Park Board 2007). This plan set out the vision and guiding principles of the restoration work.

In accordance with this plan, snags, standing live trees in damaged areas and a portion of the woody debris were retained. The amount of woody debris was reduced to between 8 to 12 kg per m². Standing live trees were spiral pruned to increase windfirmness. Trees were planted in the windthrown areas. Unlike past restoration projects where only Douglas-fir was planted, 4 native species of trees were planted. They were Sitka spruce, grand fir, western redcedar and Douglas-fir.

Standing trees were felled if they were determined to be unsafe by a danger tree assessor. A few danger trees, however, were retained if they provided high habitat value. Most trees deemed safe to leave in place were retained, but some western hemlocks that were considered safe were removed since the species was over-represented in the retained overstory. Felling was done by hand. Woody debris and felled trees were removed with a back-hoe using the hoe forwarding technique (also known as 'hoe-chucking') (Paul Lawson, R.P.F., pers. comm.). Debris on the seawall was removed by rubber-tired skidders. Machine operators were given the prescription and allowed to vary the amount, size and composition of the woody debris left on

site. Fine woody debris was chipped and hauled away for bioenergy. Logs were sold or donated for cultural projects where possible.

Trees were planted in clusters of 2 to 5 trees or individually in areas of severe windthrow where compatible with scenic view preservation (Vancouver Park Board 2009). Species composition of the planting clusters was prescribed by forestry consultants from B.A. Blackwell and Associates. Prescribed planting densities were 450 stems per ha in 6 of the 7 planting areas. In the other planting area the prescribed planting density was 250 stems per ha. Natural regeneration was expected to fill in unplanted spaces between clusters. Planting was done in 2007. In 2008, 2009 and 2010, understory vegetation was removed around planting clusters by manual brushing (Bill Stephen, pers. comm.). One or two brushing passes were made per year. In 2010, natural red alder and bitter cherry regeneration was removed where the density of these species exceeded target stand conditions (Bill Stephen, pers. comm.). Annually since 2008, Himalayan blackberry (*Rubus discolor* Weihe & Nees) germinants have been pulled in windthrown areas.

3.4 Methods

3.4.1 Historic structure and composition change

Aerial photographs from 1946, 1949, 1961, 1963, 1979, and 1994 with a scale between 1:10,000 and 1:15,000 and aerial photographs with unknown scales from 2006 and 2008 were used for analyses. The 2006 and 2008 aerial photographs were provided as rectified digital images by the Vancouver Park Board. The other photographs were from the Air Photo Collection in the Geographic Information Centre at the University of British Columbia. These

photographs were scanned at 600 dots-per-inch resolution. In some cases several frames were needed to cover the entire park. In these cases Adobe Photoshop 8.0 (Adobe Systems 2008) was used to merge the photographs to create a single mosaic covering Stanley Park. Each photograph or mosaic was then rectified in ArcGIS 9.3. Control points used to georeference the photographs were based on the 2008 GeoTIFF image of Stanley Park. Once the control points were selected, a 3rd order polynomial transformation was used to create a rectified image.

Changes in forest cover were detected by looking for changes in canopy density, roughness, or colour, when possible. Only gross changes in canopy structure or composition could be detected such as openings created after windstorms in 1962 and 2006, or development of deciduous stands in areas that once were conifer stands.

Bakewell (1980) also tracked changes from 1933 to 1971 in a report to forest managers presenting a forest maintenance program. He classified the forest in broad categories based on coniferous and/or deciduous composition and canopy openness. His findings were used in addition to those from the analysis described above to better explain changes in forest structure or composition.

Mapping of the 1962 and 2006 windthrow damage was done using aerial photographs. The damage from 1962 was mapped by Craig Farnden, R.P.F. while the mapping of 2006 damage was done by Steve Mitchell, R.P.F. Different damage thresholds were used to define severe damage in the two mapping projects. In the windthrow mapping, damage of $\geq 60\%$ of the canopy in 1962 was considered to be severe while loss of $\geq 70\%$ of the canopy in 2006 was

considered severe. Low to moderate damage was mapped in both cases in places where canopy damage was apparent up to the severe damage threshold.

3.4.2 Vegetation mapping

A UTM grid with intersections every 100 m was the basis for the systematic sampling of the forest in Stanley Park. Intersection points on the grid were randomly selected for sampling. From May to August, 2008, 182 plots were established in the forest area of the park. Of these, 56 were permanent sample plots (PSP) and 126 were temporary sample plots (TSP). Data from 179 were used in analysis. Data from 3 plots were not analyzed because of plots were later found to be outside the mapped forest boundary. These plots were located on roads or areas with turf grass. Permanent plots differed from temporary plots in that they were marked with a metal stake buried at plot centre, photographs were taken of the plots, and soil pits were dug.

In 2009, 29 additional plots (3 PSPs and 26 TSPs) were added. The plots were added in forest polygons outlining different cover types mapped in the vegetation mapping project described below. Plots were added to forest cover polygons greater than 1 ha and forest cover classes which were poorly represented in the 2008 data. The additional plots were added at the UTM grid intersection points when the points fell within an under-sampled forest type. In other cases they were placed along the UTM easting grid lines with the northing coordinate randomly selected using Microsoft Excel random number generator with limits placed so the plot fell entirely within the targeted polygon. When all of these locations were exhausted, but additional plot density was desired in a polygon, plots were added along the 100 m UTM northing grid

lines, with the easting position randomly selected as above. These plots were established in between June and August, 2009.

At each PSP and TSP, data were collected to describe the trees, large woody debris, understory vegetation, germinants, and topographic attributes of the plot (Table 3-1). Biogeoclimatic subzone and site series were keyed from topographic and soil attributes or using plant community attributes according to Green and Klinka (1994). Soil data were collected at the PSP centres including soil color, depth to C-horizon, B-horizon texture, and humus form. The azimuth and distance from plot centre to all trees in PSPs were also recorded.

Tree and understory vegetation data were collected in an 11.28 m radius plot. All tree stems taller than 1.30 m (breast height) were measured in each plot. Forks above and below breast height were noted as well as whether the tree was leaning, bent, cut, pruned, or had a broken top. Presence of wildlife activity in trees was also noted. A 1.78 m radius subplot with the same centre point as the 11.28 m plot was used to collect germinant data. Only trees that had germinated in the year of sampling were considered germinants in the 2008 survey. Germinant and seedling counts were limited to 50 individuals. If more than 50 individuals were present in a plot, '50' was recorded.

In October 2008, photographs were taken at all 2008 PSP locations facing 0, 90, 180 and 270 degrees from the plot centre, as well as vertically with a hemispheric lens. In some plots the deciduous leaves had begun to fall. Similar photographs were taken at the PSP locations added in 2009, in June.

Course woody debris was surveyed along two 20 m transects starting from the plot centre. The first transect azimuth was randomly selected. The second transect began at the plot centre point but was oriented 90 degrees clockwise manner from the first transect. Only woody debris greater than 7.5 cm in diameter at the point of intersection with the transect line and in decay classes 1 to 5 were counted. Decay classes were based on BC Ministry of Forest and Range (2007) descriptions (Fig. 3-1). Portions of transects that were on roads or trails were recorded as 'distance lost'. Volume of coarse woody debris was used to calculate mass of course woody debris per m² using the conversion factor of 333 kg per m³ found in Feller (2003).

Table 3-1. Data collected at each permanent and temporary sample plot.

Tree data	Germinant data	Coarse woody debris data	Understory vegetation data	Physical location
Species	Count by species. Counts greater than 50 recorded as 50	Species	Percent cover bryophytes by species if greater than 1%	Slope and aspect
Diameter at breast height		Intersect diameter	Percent cover and height of vegetation by species if greater than 1% (for invasive species percent cover only)	Vegetation cover (%)
Height (for selected trees)		Bottom end diameter	Count and average height of natural tree regeneration under 1.3 m and over 2 years old. Counts greater than 50 recorded as 50	Trail areas (%)
Height to bases of live crown (for selected trees)		Top end diameter	Count and average height of planted trees.	Road areas (%)
Decay class following BC Ministry of Forest and Range (2007)		Condition of top end (e.g. cut, broken, etc.)	Average growth of planted and naturally regenerated Douglas fir under 1.3 m tall in preceding year	Windthrow area (%)
		Condition of bottom end		Number of pits
		Length of piece		Site series (up to 2 entries)
				Basal area using a prism
				Soil colour (permanent plots only)

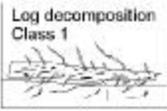
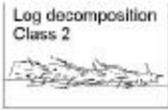
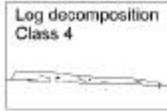
	 Log decomposition Class 1	 Log decomposition Class 2	 Log decomposition Class 3	 Log decomposition Class 4	 Log decomposition Class 5
	CLASS 1	CLASS 2	CLASS 3	CLASS 4	CLASS 5
WOOD TEXTURE	intact, hard	intact, hard to partly decaying	hard, large pieces, partly decaying	small, blocky pieces	many small pieces, soft portions
Other associated characteristics					
PORTION ON GROUND	elevated on support points	elevated but sagging slightly	sagging near ground, or broken	all of log on ground, sinking	all of log on ground, partly sunken
TWIGS < 3 cm (if originally present)	twigs present	no twigs	no twigs	no twigs	no twigs
BARK	bark intact	intact or partly missing	trace bark	no bark	no bark
SHAPE	round	round	round	round to oval	oval
INVADING ROOTS	none	none	in sapwood	in heartwood	in heartwood

Figure 3-1. Large woody debris classification (from BC Ministry of Forests and Range 1997).

To better understand post-windthrow regeneration dynamics, additional data were collected to determine the stocking density of regeneration in windthrow areas in Stanley Park. As described in Chapter 4, these data were used for modeling scenarios to project future stand conditions at Stanley Park.

TSPs and PSPs located in areas mapped by Steve Mitchell, R.P.F., as having low to moderate or severe windthrow were sampled. Severe damage was defined as $\geq 60\%$ canopy loss. Lower to moderate damage was defined as $< 60\%$ canopy loss. Areas where windthrow did not occur were not sampled because regeneration, particularly regeneration since 2006, was not expected in those areas. Any regeneration in undamaged areas was also not expected to attain overstory size because of the generally closed canopy found outside of windthrow areas. Future disturbance will be required for the regeneration in these to reach the overstory.

The planted tree and seedling plots were 11.28 m in radius. The plot centres for the planted tree and seedling plots were the same as those used in vegetation survey plots installed in 2008 and 2009. The number of naturally regenerated individuals by species > 30 cm and ≤ 1.30 m tall was recorded in the seedling category.

Trees planted in 2007 as part of the restoration effort were counted separately. Planted trees were identified by their location and growth form. Planted trees were located in clumps spanning several metres near painted bamboo stakes. Planted trees typically had vigorous first and second year growth resulting from growing in a tree nursery. Later growth, which occurred after planting in Stanley Park, was variable. All planted trees were the same age (4 years old in 2009).

Data collected on planted trees included diameter 10 cm above the ground, total height at time of sampling, height at the end of the previous growing season (for all species except western redcedar), tree condition, and height relative to surrounding vegetation within a 1 m

radius. Whether the tree was unobstructed without overhead or significant side shading from neighbouring plants was also recorded. In some cases brushing crews had cleared the area around planted trees making them much taller than the surrounding vegetation. Brushing was not recorded because the timing of brushing could not be determined (i.e. it was difficult to tell if brushing had occurred at the end of the previous growing season or the beginning of the current growing season). A tree could still be deemed obstructed if it was estimated that the remaining vegetation could grow to over top the tree in one growing season or less.

Trees from 0 to ≤ 30 cm tall were considered germinants and there was no maximum count number. This is different than the protocol described above. The protocol for germinants was changed to better understand the density of regeneration in the smallest size classes. In each plot, 1 of the 5 germinant subplots sampled at each location was centered at that plot centre. The 4 other subplots were centered 11.28 m in each of 4 cardinal directions (0, 90, 180, and 270 degrees) (Fig. 3-2). Counts of germinants by species were done for 3 height classes, 0 to ≤ 10 cm, >10 to ≤ 20 and > 20 to ≤ 30 cm. Counts were done in 5 subplots at each sampling location and the numbers were not initially pooled. For analysis, data were pooled by plot.

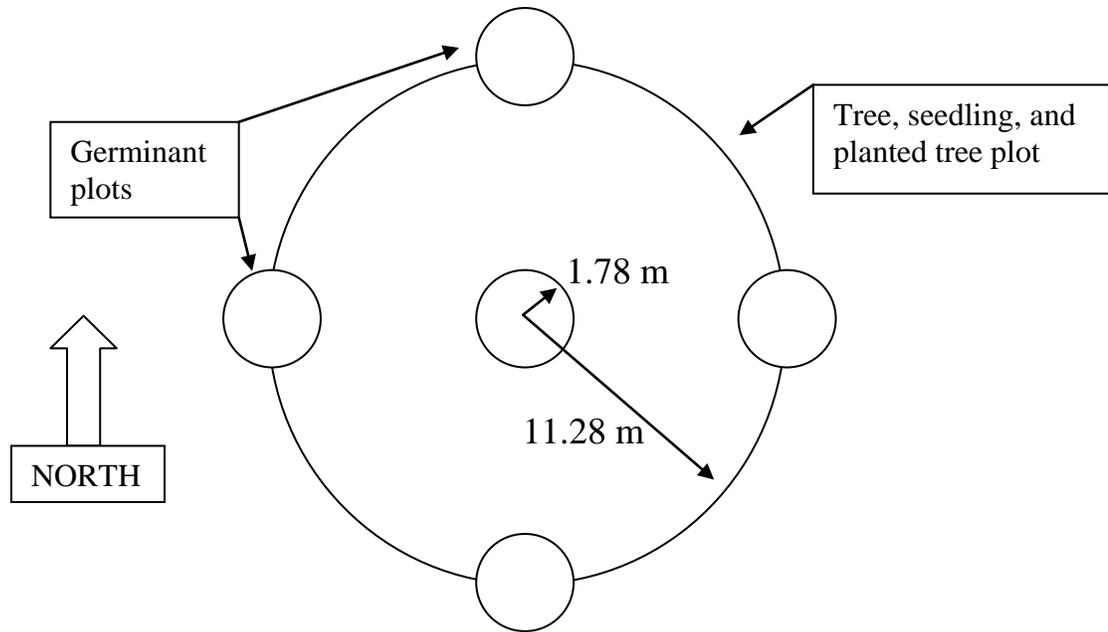


Figure 3-2. Diagram of tree, seedling, and planted tree sampling plot and 5 germinant plots, 1 at plot centre and 1 at each cardinal direction.

Understory vegetation composition and cover was estimated to the nearest 5 percent with additional categories for 1 and 3 percent. Percent cover data and average height of understory species were collected by species, if they represented $> 1\%$ of the 11.28 m plot area. Percent cover of bryophytes was also recorded by species if they covered $> 1\%$ of the plot area.

3.4.3 Vegetation mapping

Beese and Paris (1989) produced forest vegetation and site series maps of Stanley Park. Their maps were created as part of a proposed management plan for the Park. These maps were used as the basis for the vegetation mapping project described here.

Other map layers used in the mapping project showed Freda plantations, small plantations initiated between 1989 and 2002, 2008 planting blocks, and moderate and severe

windthrow areas. Freda plantations were digitized from an aerial photograph on which Craig Farnden, R.P.F., marked plantation polygons. The Vancouver Park Board provided the layer showing plantations initiated between 1989 and 2002. The plantations were installed to fill canopy gaps (Bill Stephen, pers. comm.). The windthrow area layers were provided by Steve Mitchell, R.P.F. Finally, the layer of 2008 planting blocks was provided by B.A. Blackwell and Associates. The 2008 planting blocks were drawn by staff from B.A. Blackwell and Associates. Each block had an associated planting prescription.

The Beese and Paris (1989) map layers were overlaid with the other map layers in ArcView 3.3 (ESRI 2002). The 1989 forest vegetation polygons were then split by all the other layers. This created over 1,000 forest type polygons, each outlining an area with a unique history of disturbance and management within each site series. Subsequent work was done in ArcGIS 9.3 (ESRI 2009).

Where adjacent polygons had similar vegetative conditions they were merged together. The vegetative conditions were determined by comparing the data from survey plots in the polygons and comparing the canopy structure in the high-resolution (15 cm resolution) 2008 aerial photograph. In March and April 2009, transects along lines of sample points were surveyed to determine changes in site series and canopy structure and to finalize the simplified vegetation type mapping. Boundaries were recorded on printed aerial photographs taken in 2008. Final edits were made to match polygon boundaries with apparent boundaries on the aerial photograph taken in 2008.

3.4.4 Data analysis

All vegetation, regeneration and site data were initially entered into Excel. These data were then imported into SAS Version 9.2 statistical software (SAS Institute 2003) for analysis. Results were exported from SAS to Excel to prepare graphs for presentation.

The vertical hemispherical photographs taken at each permanent sample plot were analysed with Gap Light Analyzer (GLA) 2.0 (Frazer et al. 1999). GLA determines the maximum possible light transmission of both direct and diffuse light based on local latitude, longitude, altitude, slope and aspect. GLA then calculates light gap transmission of both direct and diffuse light (in Mols per m² per day and as a percent to maximum light transmission) and provides estimates of leaf area index. Default settings were used for all factors with the exception of site settings and growing season length (Table 3-2). Analyses were based on percent total light transmission and were done in SAS.

Table 3-2. GLA settings used for hemispherical photograph analysis which were modified from default settings.

Factor	Input
Location	Centre of sampling grid used for all analysis. Latitude: 49° 18' 12" Longitude: 123° 8' 28"
Elevation	Elevation of 1 m contour line closest to plot center taken from GIS
Orientation	Slope and aspect taken from survey data for each plot
Growing season length	Start date: April 15 End data: October 15

Statistical analysis was done using SAS. Analysis of variance (ANOVA) was used to compare the total plot basal area in the first sampling year (2008 or 2009 depending on the plot)

by vegetation class and age class (Table 3-3) and in a different ANOVA by vegetation type (the combination of vegetation and age class used in mapping) (Table 3-4). Scheffé’s post-hoc test was used for pair-wise comparisons. Scheffé’s post-hoc test was selected because it can be used to compare means of groups with different sample sizes (Sit 1995). Alpha was 0.05 of all tests.

Table 3-3. ANOVA table for test of basal area by vegetation type.

Source	Degrees of freedom	Expected mean squares	F-ratio
Vegetation type (A)	12	$\sigma_{\epsilon}^2 + \phi_A = MS(A)$	MS(A)/ MS(E)
Error	195	$\sigma_{\epsilon}^2 = MS(E)$	
Total	207		

Table 3-4. ANOVA table for test of basal area by vegetation class and age class.

Source	Degrees of freedom	Expected mean squares	F-ratio
Vegetation class(A)	4	$\sigma_{\epsilon}^2 + \phi_A = MS(A)$	MS(A)/MS(E)
Age class(B)	2	$\sigma_{\epsilon}^2 + \phi_B = MS(B)$	MS(B)/MS(E)
Error	201	$\sigma_{\epsilon}^2 = MS(E)$	
Total	207		

3.5 Results

3.5.1 Historic structural changes

Changes in the canopy structure between 1933 and 1946 can be inferred from comparing Bakewell’s (1980) map of canopy cover in 1933 and the 1946 aerial photograph (Figs.3-3 and 3-4). The photograph from 1946 shows a more open canopy in the southeast quadrant of the park and west of Beaver Lake. Bakewell notes that between 1933 and 1952, the next year of his

photo-analysis, deciduous and coniferous canopy type increases greatly. These changes in canopy cover can possibly be attributed to the 1934 windstorm.

From 1946 to 1961 the canopy increased greatly in density, particularly in the section of park between Beaver Lake and Prospect Point (Figs 3-4, 3-5, and 3-6). Areas that were densely forested in 1946 remain densely forested in the later image. The 1962 storm caused substantial damage in Stanley Park. In contrast to the closed canopy forest of 1961 (Fig. 3-6), the aerial photograph from 1963 (Fig. 3-7) shows a wide swath of open canopy extending from the southwest to the northeast, as well as significant reductions in tree density south of Beaver Lake and on Brockton Point.

After the 1962 storm debris was cleared throughout the park and planting sites were created (Bill Stephen pers. comm.). Douglas-fir seedlings were planted at high densities in the newly open areas, particularly on west side of the park. In the aerial photographs from 1979 (Fig. 3-8) dense canopies of young trees are visible in many areas that were open in the 1963 photograph.

The canopy appears closed in most areas in the 1994 aerial photograph and in the aerial photograph taken in 2006 prior to the December windstorm (Figs. 3-9 and 3-10). The exception appears to be the stand south of Beaver Lake. This stand may have established following the 1885 fire and is less dense in 2006 than in earlier photographs. This reduction in canopy density is possibly a result of natural stand development. As stands mature it is typical for crowns to

become well spaced and gaps to begin forming (Oliver and Larson 1996). The winter storm later in 2006 changes the appearance of the canopy greatly.

The 2008 aerial photograph shows that large openings were formed near Prospect Point, north of Lost Lagoon, and east of Beaver Lake (Fig. 3-11). Most of these areas were not damaged in 1962, but a few areas were damaged in both storms (Fig. 3-12).

Mapping of damage from the 1962 and 2006 storms showed that more area was damaged in 2006 than in 1962. In 1962, 23 ha were severely damaged ($> 70\%$ loss of canopy) while 33 ha had moderately or low level damaged ($\leq 70\%$ loss of canopy). In 2006, 31 ha were severely damaged ($> 60\%$ loss of canopy) and 50 ha had moderately or low level damaged ($\leq 60\%$ loss of canopy).

3.5.2 Composition changes

Changes in species composition are more difficult to evaluate from aerial photographs than changes in canopy closure and stand maturity. But some changes in composition are apparent on the photographs and were documented by Bakewell (1980) and others. Bakewell's (1980) maps show an increase in deciduous cover from 1933 to 1978, which he attributes to disturbance and the death of large diameter coniferous trees. For example, the stand between Beaver Lake and the Rose Garden appears to be coniferous in 1946 but regenerated as a deciduous stand following Hurricane Freda.

One anecdote on species composition that has had a particular impact on the management of Stanley Park is that when the park was created, it is said, western hemlock, western redcedar, and Douglas-fir were found in nearly equal proportion. This has led to ongoing concern over the lack of Douglas-fir regeneration. Kheraj (2007) notes that Malcolm Swine, the entomologist responsible for the program to combat insect outbreaks in early 20th century, suggested that western hemlock be replaced with Douglas-fir to create a healthier forest in light of hemlock looper outbreaks. A summary of forestry reports from 1910 to 1938 (Vancouver Park Board 1939) notes that in 1914 a forester reported disparagingly that areas of second growth, except where cleared, were covered by tangled, almost impenetrable, low growth of western hemlock, western redcedar, red alder, and salmonberry (*Rubus spectabilis* Prush), and much dead and down debris. Bakewell (1980) commented that western hemlock had become the dominant species due to its high shade tolerance. He goes on to prescribe clearing of understory vegetation and planting Douglas-fir in many areas in the park. Bakewell (Dave Bakewell, R.P.F., pers. comm.) also commented that whenever possible western hemlock was removed from the understory during routine forestry operations in the park.

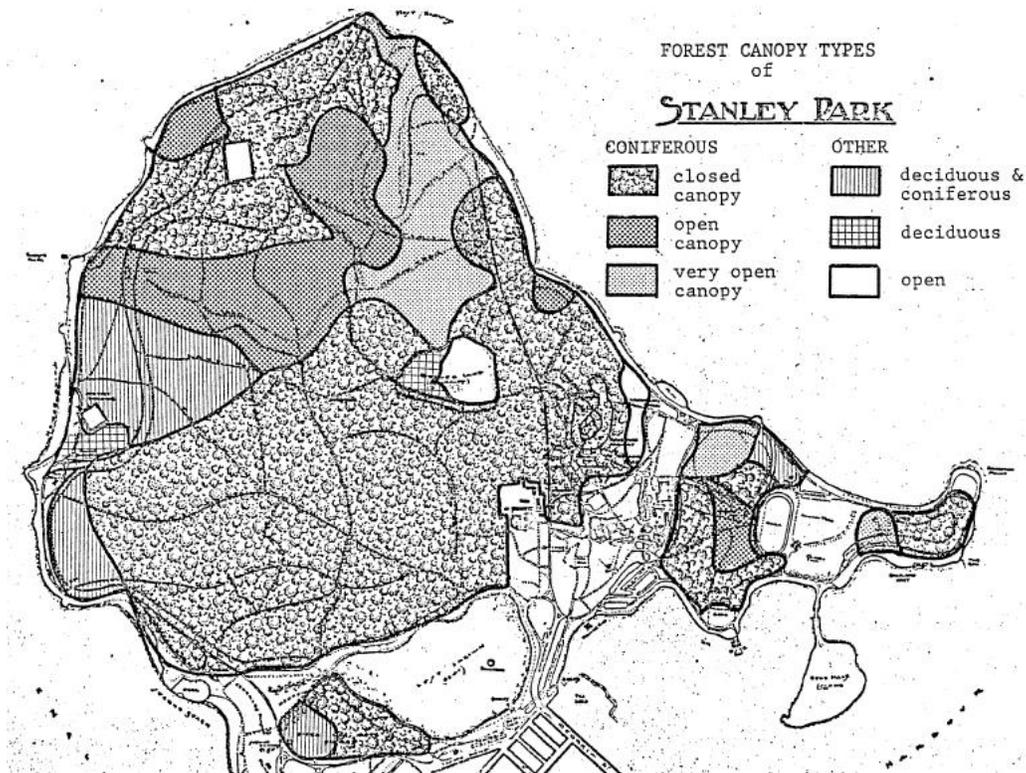


Figure 3-3. Map of forest cover types in Stanley Park in 1933 (from Bakewell 1933).

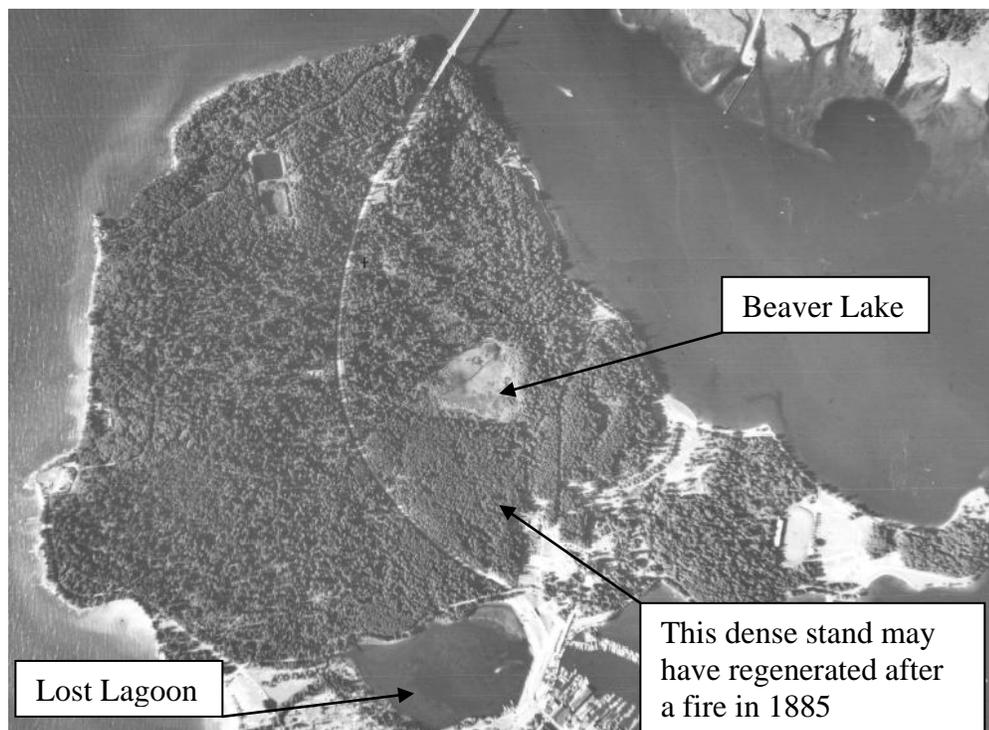


Figure 3-4. 1946 aerial photographs of Stanley Park.



Figure 3-5. Mosaic of 1949 aerial photographs of Stanley Park. Lighting changes and possibly the season the photograph was taken are likely responsible for the more open canopy appearance as compared to the 1946 aerial photograph.



Figure 3-6. Aerial photograph of Stanley Park taken in 1961.



Figure 3-7. Mosaic of aerial photographs of Stanley Park taken in 1963.

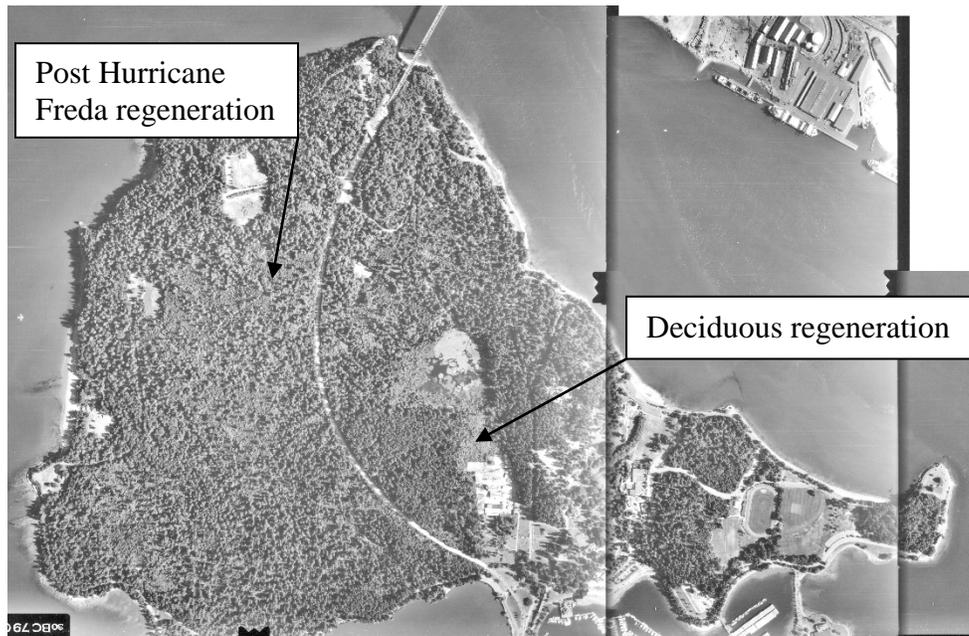


Figure 3-8. Mosaic of aerial photographs of Stanley Park taken in 1979.

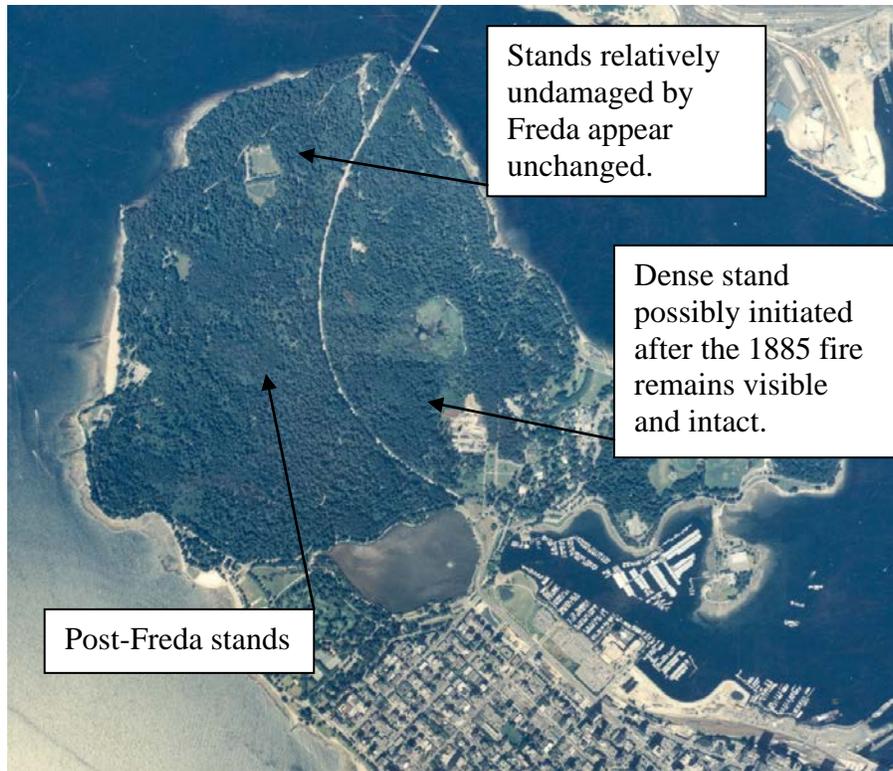


Figure 3-9. Aerial photograph of Stanley Park taken in 1994

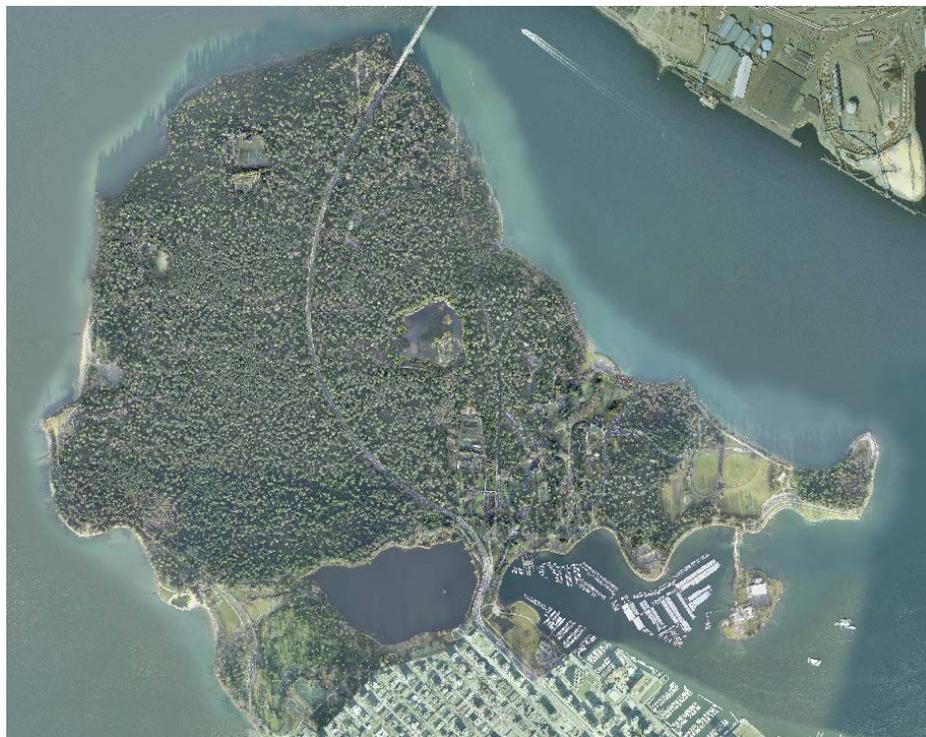


Figure 3-10. Aerial photograph of Stanley Park taken in 2006



Figure 3-11. Aerial photograph of Stanley Park taken in 2008

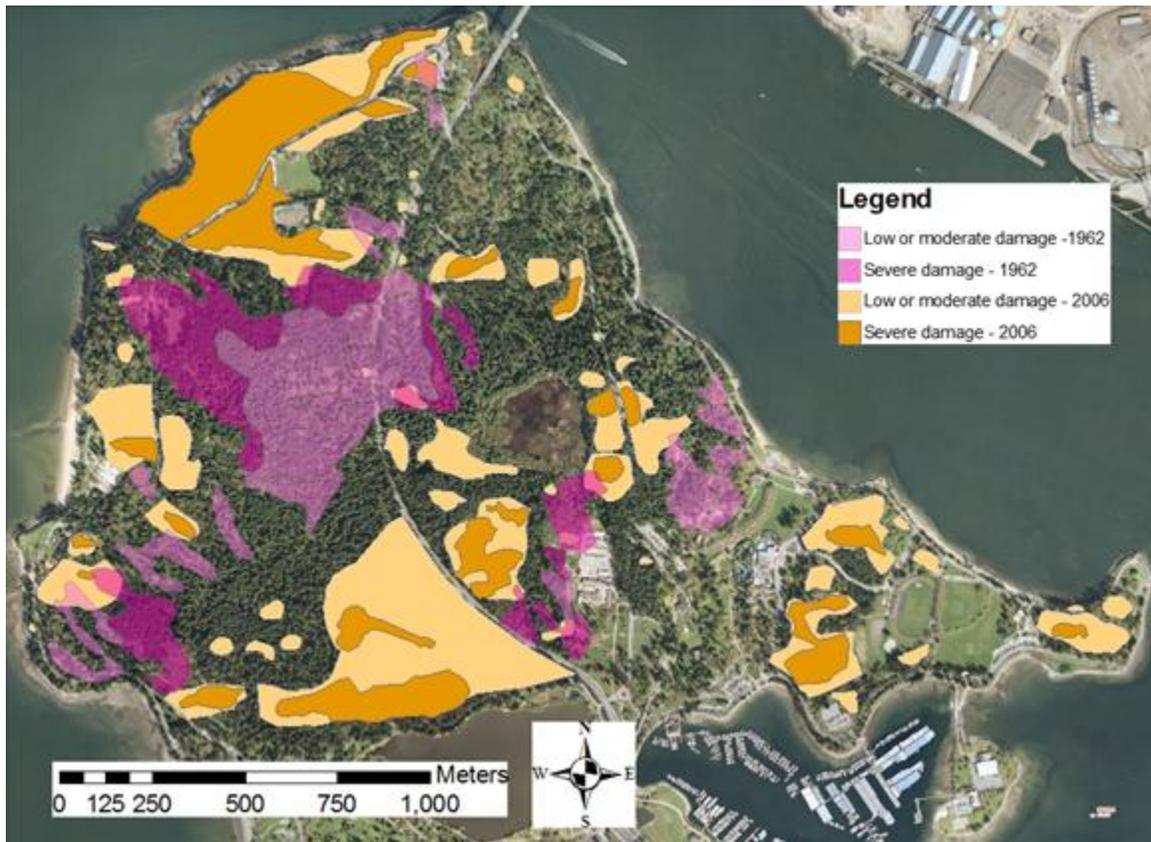


Figure 3-12. Severe and low or moderate windthrow damage from 1962 and 2006 windstorms. Loss of > 70% of the canopy in 1962 was considered to be severe, while loss of > 60% of the canopy in 2006 was considered severe. Low to moderate damage was mapped in both cases in places where canopy damage was apparent up to the severe damage threshold. Mapping of the damage from 1962 was done by Craig Farnden, R.P.F. Mapping of the damage from 2006 was done by Steve Mitchell, R.P.F.

3.5.3 Vegetation types

All plots in Stanley Park fell within the CWHdm and CWHxm subzones as defined in Green and Klinka (1994). Within these subzones, most plots were classified as either site series 05 or 07, but site series 01, 06, and 12 were found as well (Fig. 3-13). For vegetation mapping, all CWHxm stands were considered part of one vegetation type because of the small extent of this subzone at Stanley Park. The coniferous CWHdm plots were initially divided into wet and

dry forest classes. The dry coniferous class included site series 01 and 05, the dry coniferous class included site series 06, 07 and 12. Some plots had high basal areas of red alder or bigleaf maple (*Acer macrophyllum* Prush). These plots were separated into two classes called alder dominated and mixed coniferous/deciduous.

Age data were not collected in the field during this vegetation survey. Nevertheless, it was possible to determine the approximate age of some stands. Beese and Paris (1989) provide age data for stands they mapped, while the age of Freda plantations, 1989 to 2002, and 2006 plantations was known. Using these data the age of each stand was estimated. Stands were then assigned to 1 of 3 age classes, young (0 to 20 years), mature (20 to 100 years) and old mature (> 100 years). These ages refer to age of the canopy trees in the stand, not the oldest or largest trees.

The combination of an age class and vegetation class was termed a vegetation type. Under this classification system 15 possible vegetation types were possible (Table 3-6). Of these, 13 were found in Stanley Park. No stand was classified as either young alder or mature CWHxm.

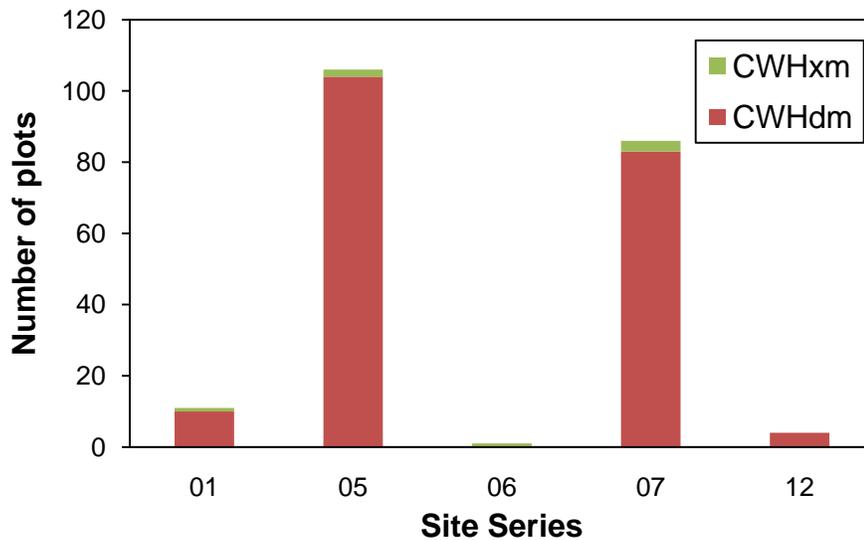


Figure 3-13. Site series of plots in Stanley Park by biogeoclimatic subzone.

Table 3-5. Vegetation and age classes used in vegetation mapping. Vegetation type was taken to be the combination of one vegetation class and one age class.

Vegetation class	Age classes
Wet conifer (CWHdm 07, 12)	Young (0 – 20 y)
Dry conifer (CWHdm 01, 06, 05)	Mature (> 20 – 100 y)
Alder stands (CWHdm 05, 07)	Old Mature (> 100 y)
Mixed coniferous/deciduous stands (CWHdm 07)	
Very dry coastal (CWHxm)	

3.5.4 Tree species composition

Individuals > 1.30 m tall of fifteen different trees species were found in the forests of Stanley Park. Western hemlock was the most common species with an average of 255 trees per hectare (Fig. 3-14). The five most frequently found species, western hemlock, western red cedar, Douglas-fir, red alder and bigleaf maple, are the focus of further analysis of the overstory composition. The relative dominance of these species in the overstory was used to divide stands

into five vegetation classes. All other species are included in the “other” category in analysis (Table 3-7). Sitka mountain-ash (*Sorbus sitchensis* Roemer), the sixth most common species, was grouped in the ‘other’ category because it does not typically attain canopy status, even though it is nearly 5 times as common as the next most common of the remaining species.

Less common native tree species are grand fir, Sitka spruce, Sitka mountain-ash, Pacific crabapple (*Pyrus fusca* Raf.), paper birch (*Betula papyrifera* Marsh.), bitter cherry (*Prunus emarginata* (Dougl.) Walp.) and black cottonwood (*Populus trichocarpa* T. & G.). Seedlings of arbutus (*Arbutus menziesii* Pursh.) and western white pine (*Pinus monticola* Dougl.) were seen in the inventory in 2008 and 2009. Numerous exotic tree species have been planted or have naturally regenerated in Stanley Park. A few were found in the forest area including white oak (*Quercus alba* L.), Norway maple (*Acer platanoides* L.), horse-chestnut (*Aesculus hippocastanum* L.), and chestnut (*Castanea spp.*).

A few species poorly represented in the park-wide tree category are locally important. Pacific crabapple for example is very common around Beaver Lake, but is rarely found elsewhere.

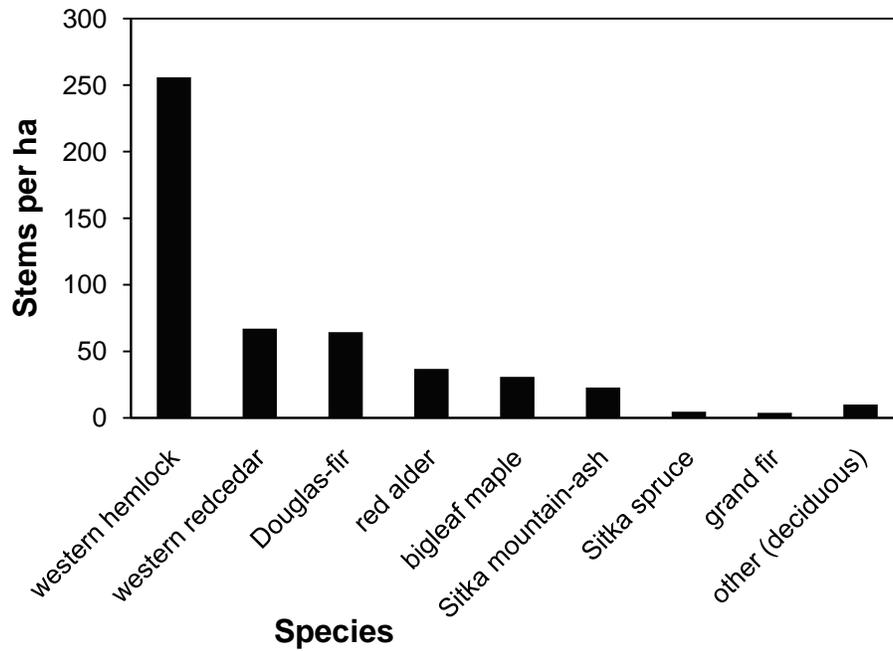


Figure 3-14. Average density of trees > 1.30 m in height by species in Stanley Park. The ‘other (deciduous)’ category includes Pacific crabapple, bitter cherry, paper birch, water birch, chestnut, horse-chestnut and white oak.

Table 3-6. Common and scientific names of all tree species found in sample plots in 2008 and 2009. Naming convention follows Hitchcock and Cronquist (1976).

Common name	Scientific name
arbutus	<i>Arbutus menziesii</i> Pursh.
bigleaf maple	<i>Acer macrophyllum</i> Prush.
bitter cherry	<i>Prunus emarginata</i> (Dougl.) Walp.
black cottonwood	<i>Populus trichocarpa</i> T. & G.
chestnut	<i>Castanea spp.</i>
Douglas-fir	<i>Pseudotsuga menziesii</i> (Mirbel) Franco.
grand fir	<i>Abies grandis</i> (Dougl.) Forbes
horse-chestnut	<i>Aesculus hippocastanum</i> L.
Norway maple	<i>Acer platanoides</i> L.
Pacific crabapple	<i>Pyrus fusca</i> Raf.
Pacific yew	<i>Taxus brevifolia</i> Nutt.
paper birch	<i>Betula papyrifera</i> Marsh.
red alder	<i>Alnus rubra</i> Bong.
Sitka mountain-ash	<i>Sorbus sitchensis</i> Roemer
Sitka spruce	<i>Picea sitchensis</i> (Bong.) Carr.
western hemlock	<i>Tsuga heterophylla</i> (Raf.) Sarg.
western redcedar	<i>Thuja plicata</i> Donn.
western white pine	<i>Pinus monticola</i> Dougl.
white oak	<i>Quercus alba</i> L.

3.5.5 Basal area

The young vegetation types have basal areas at least half as large as the mature and old mature vegetation types. Trees that survived the 2006 windthrow event make up this basal area. The western redcedar basal area in the young conifer type is composed of large diameter veteran trees (Fig. 3-15). The alder and mixed deciduous and coniferous stand types were dominated by alder and bigleaf maple, respectively, in terms of basal area.

When the total plot basal area was analysed by vegetation class and age class both were found to have a significant effect on basal area (age class $p < 0.0143$; vegetation class $p < 0.0034$). The post-hoc Scheffé's test showed that there were significant differences between the 'young' and 'mature', and 'young' and 'old mature' age classes. However, no pair-wise differences were found by vegetation class.

The analysis of variance of total plot basal area by vegetation type (the combination of vegetation and age class) showed significant differences between types ($p < 0.0121$) however, Scheffé's pair-wise analysis failed to show any significant differences. In addition, the basal areas of each of the 5 most common tree species were also analyzed by vegetation type. Significant differences were found in the basal area of each species tested except western redcedar (ANOVA, Dr $p < 0.0001$; Fd $p < 0.0001$; Hw $p < 0.0013$; Mb $p < 0.0006$). Scheffé's post-hoc test showed no difference in basal area of western hemlock between any of the vegetation types but significant differences were found for the other species (Table 3-8).

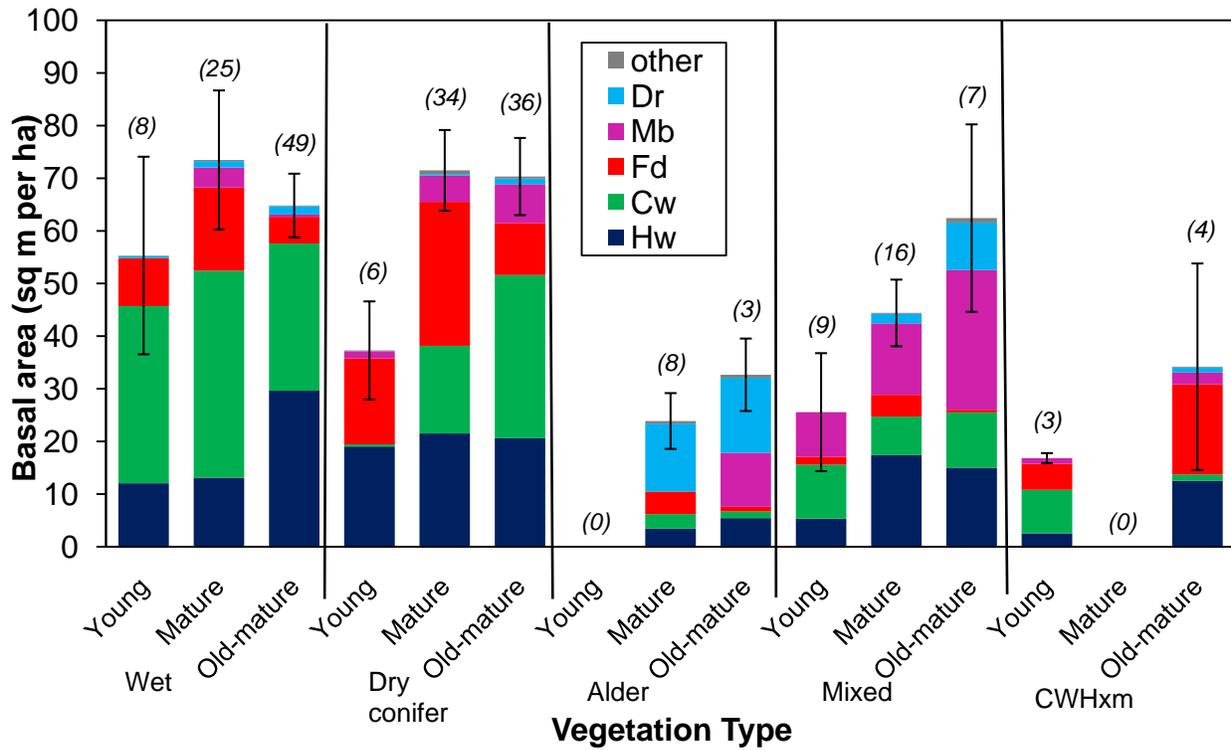


Figure 3-15. Basal area by species for each vegetation type. The numbers in parenthesis above each bar indicates number of plots. Bars on columns show one standard error of total plot basal area per ha. Dr – red alder, Mb – bigleaf maple, Fd – Douglas-fir, Cw – western redcedar, Hw – western hemlock, other – all other tree species found in Stanley Park

Table 3-7. Results of the Scheffé's test of the total plot basal areas of Douglas-fir (Fd), bigleaf maple (Mb) and red alder (Dr) by vegetation type. The species code in a cell represents a significant difference between the two vegetation types for the indicated species.

	Vegetation type*																										
Veg. type*	1A	1B	1C	2A	2B	2C	3A	3B	3C	4A	4B	4C	5A	5B	5C												
1A																											
1B																											
1C																											
2A																											
2B																Fd											
2C																											
3A																type not present in Stanley Park											
3B																Dr	Dr	Dr	Dr	Dr	Dr						
3C																Dr	Dr	Dr	Dr	Dr	Dr						
4A																						Dr	Dr				
4B																						Dr	Dr				
4C																		Mb		Dr	Dr						
5A																											
5B																type not present in Stanley Park											
5C																							Dr				

* Vegetation type codes are as follows:

1A – Young wet conifer	2A – Young dry conifer	3A – Young alder	4A – Young mixed coniferous/deciduous	1A – Young CWHxm
1B – Mature wet conifer	2B – Mature dry conifer	3B – Mature alder	4B – Mature mixed coniferous/deciduous	1B – Mature CWHxm
1C – Old mature wet conifer	2C – Old mature dry conifer	3C – Old mature alder	4C – Old mixed coniferous/deciduous	1C – Old mature CWHxm

3.5.6 Stems per hectare

Western hemlock is the most common species in the coniferous vegetation types (i.e. wet conifer, dry conifer, and CWHxm) (Fig. 3-16). In the deciduous vegetation types western hemlock is less common. Among deciduous vegetation types western hemlock is the most common species in only the mature and old mature mixed deciduous/coniferous vegetation types. In the other deciduous vegetation types red alder is most common.

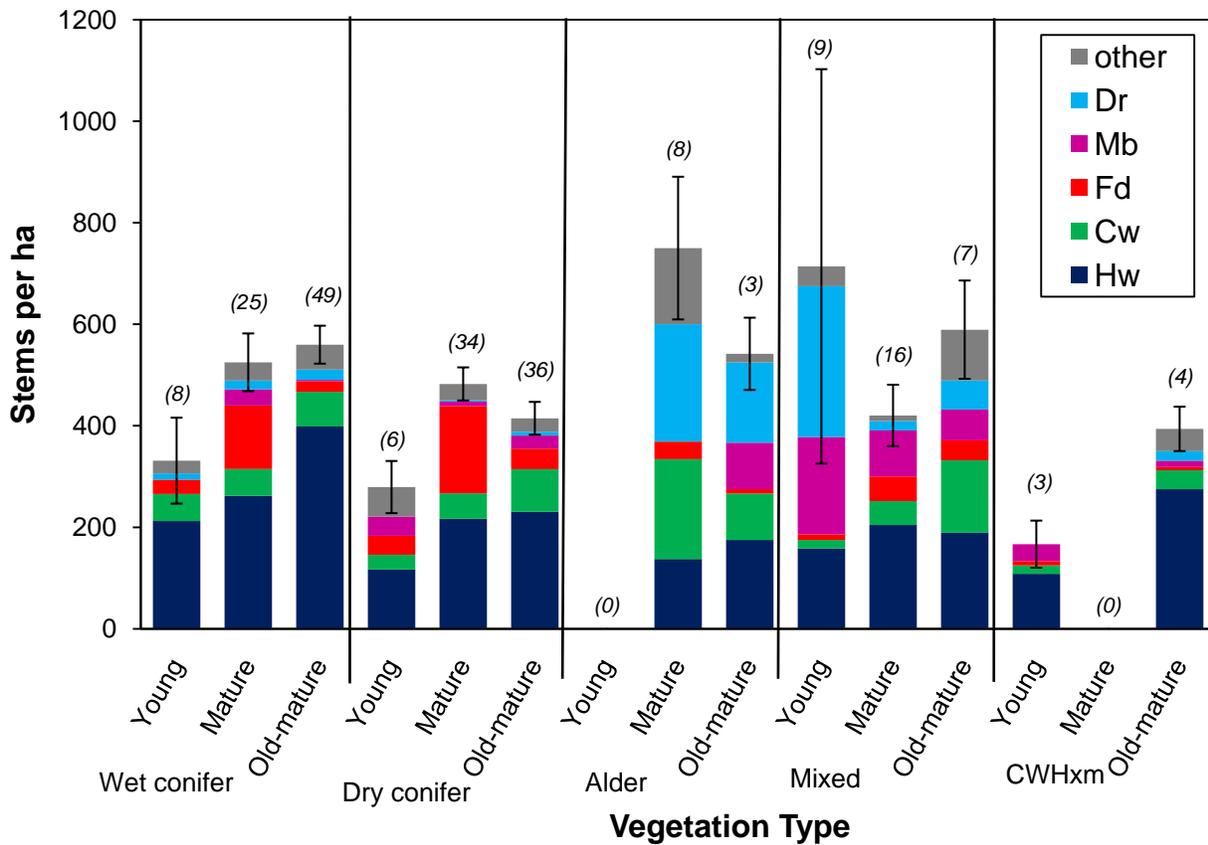


Figure 3-16. Stems per hectare by species for each vegetation type. The numbers in parenthesis above each bar indicates number of plots. Bars on columns show one standard error total of plot stems per ha. Dr – red alder, Mb – bigleaf maple, Fd – Douglas-fir, Cw – western redcedar, Hw – western hemlock, other – all other tree species found in Stanley Park

3.5.7 Diameter distributions

All vegetation types except the old mature CWHxm type show a reverse-J diameter distribution (Fig. 3-17 a-m). In the CWHxm the diameter classes create a negative linear distribution. The young dry conifer diameter distribution is generally reverse J-shaped, but a slight increase in frequency was found in the 70 and 90 cm DBH classes.

Shade tolerant species were the most common species in the smallest size class (0 to 20 cm DBH) in the wet and dry conifer types. The shade tolerant western hemlock dominates this size class in both stems per ha and basal area per ha, while western redcedar was second most common (Fig. 3-17). The 30, 50, 90 and 110 cm DBH classes are dominated in terms of stems per ha by Douglas-fir, with western hemlock second in these vegetation types. In the 70 cm DBH however, the rankings are reversed. The trees larger than 100 cm in these vegetation types are likely veterans which survived both the 1962 and 2006 storms.

A pattern of basal area distribution varied by vegetation type (Fig. 3-18 a-m). Western redcedar with DBHs > 160 cm account for the most basal area in 1 DBH class in 5 of the vegetation types. In 1 vegetation type most of the basal area is in Douglas-fir > 160 cm DBH.

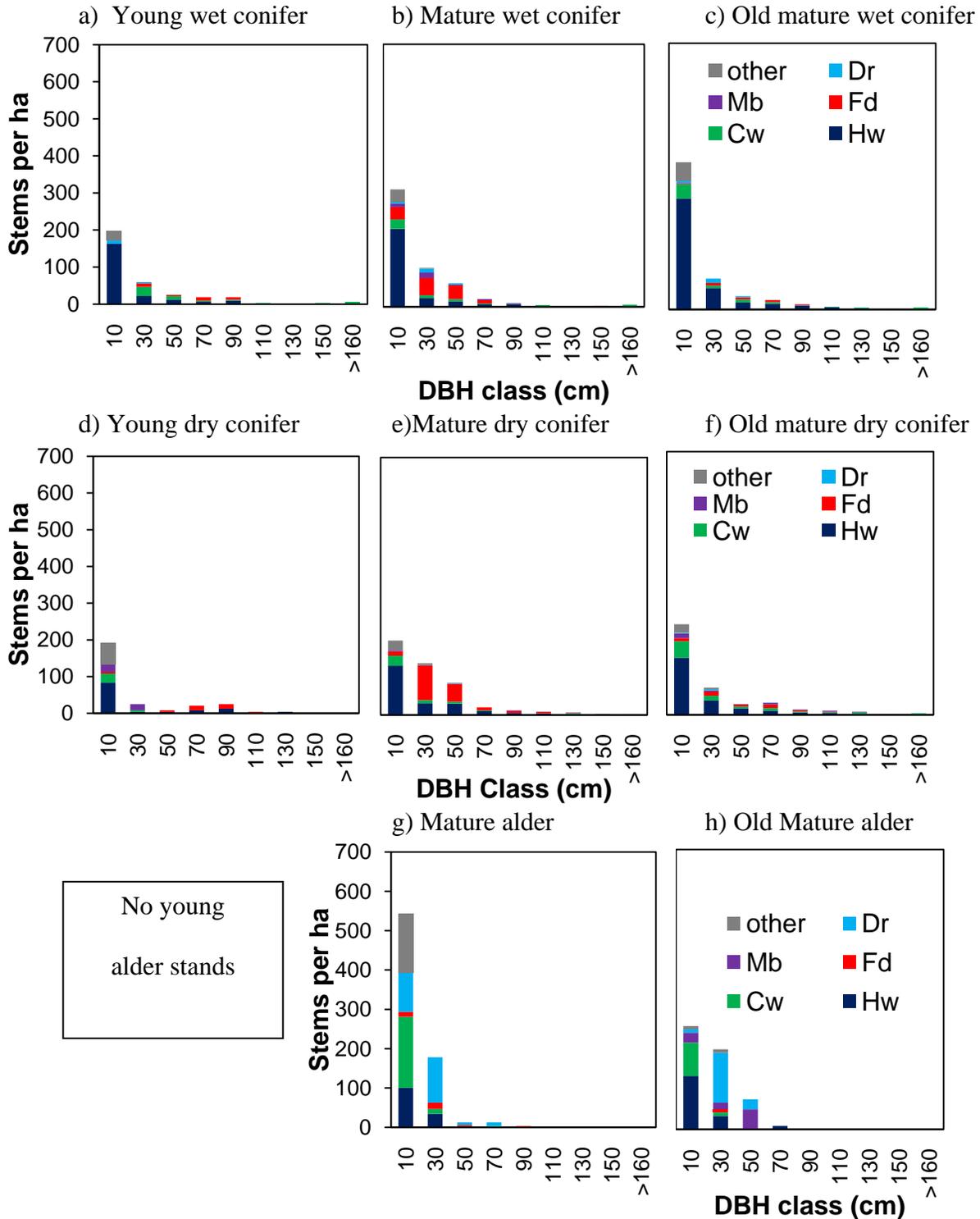


Figure 3-17. Stems per hectare for each vegetation type by DBH class. The horizontal axis for all charts displays DBH class (cm). Mb – bigleaf maple, Cw, western redcedar, Dr – red alder, Fd – Douglas-fir, Hw – western hemlock, other includes all other tree species found in Stanley Park.

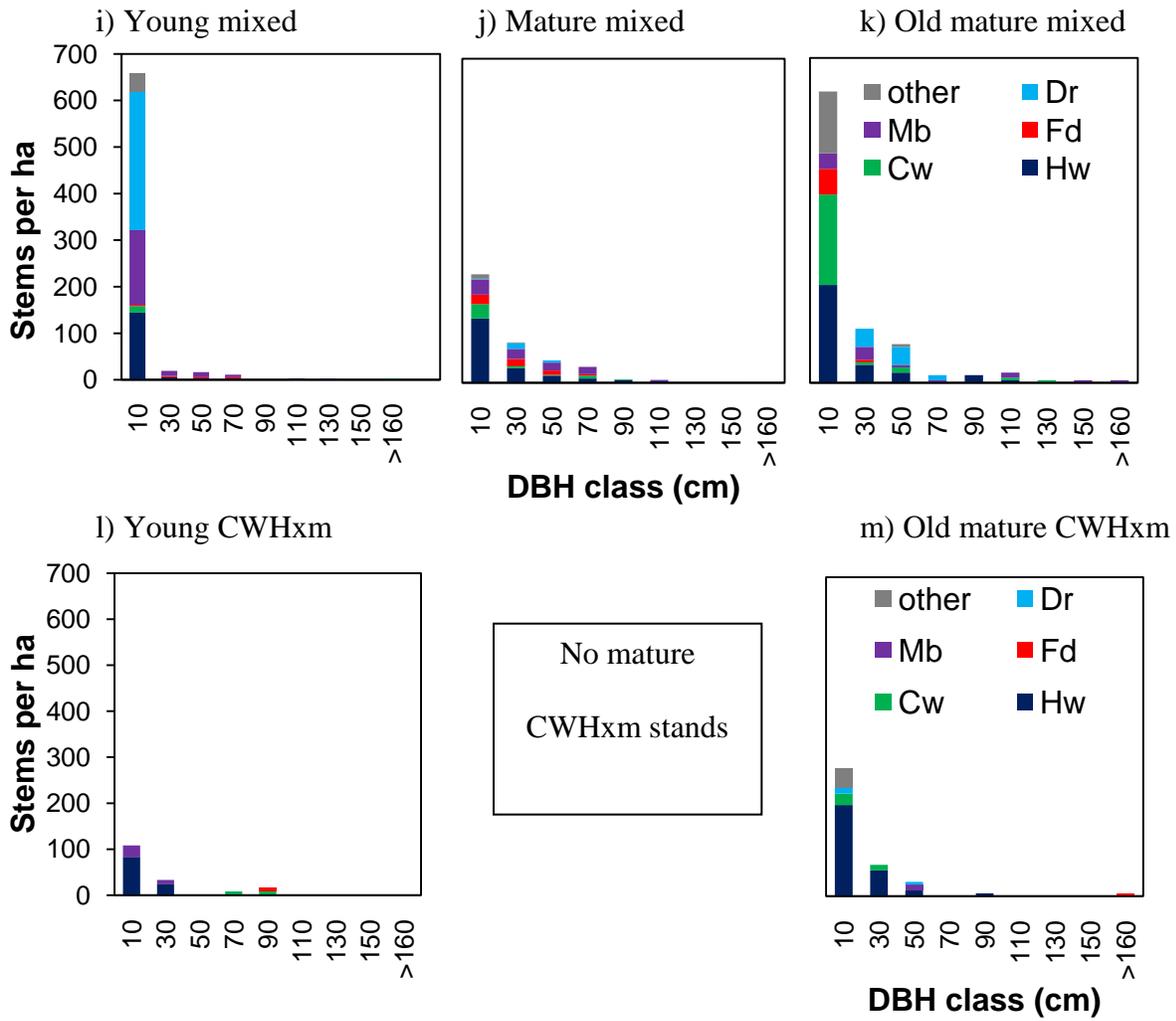


Figure 3-17. cont.

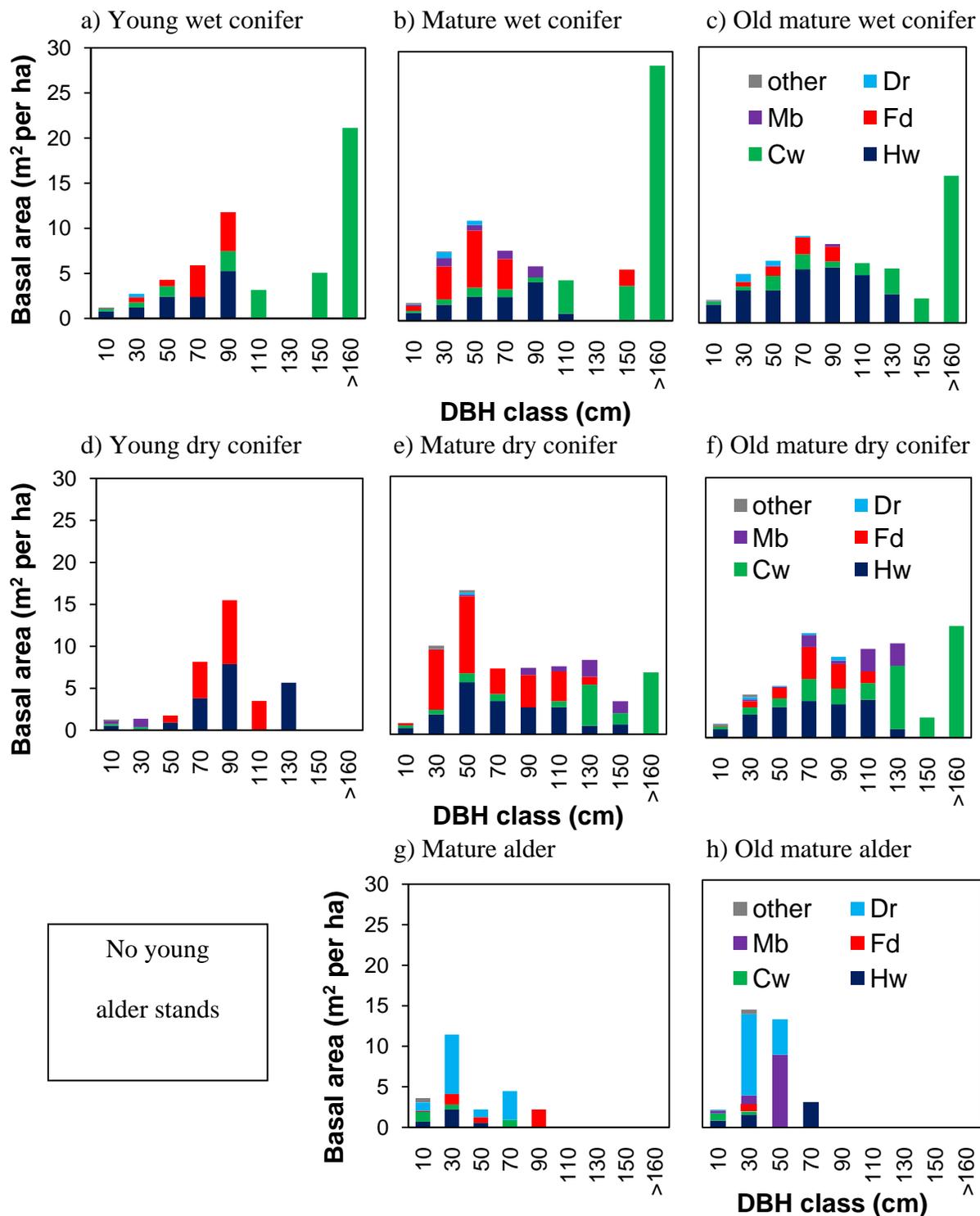


Figure 3-18. Basal area per hectare for each vegetation type by DBH class. The horizontal axis for all charts displays DBH class (cm). Mb – bigleaf maple, Cw, western redcedar, Dr – red alder, Fd – Douglas-fir, Hw – western hemlock, other includes all other tree species found in Stanley Park.

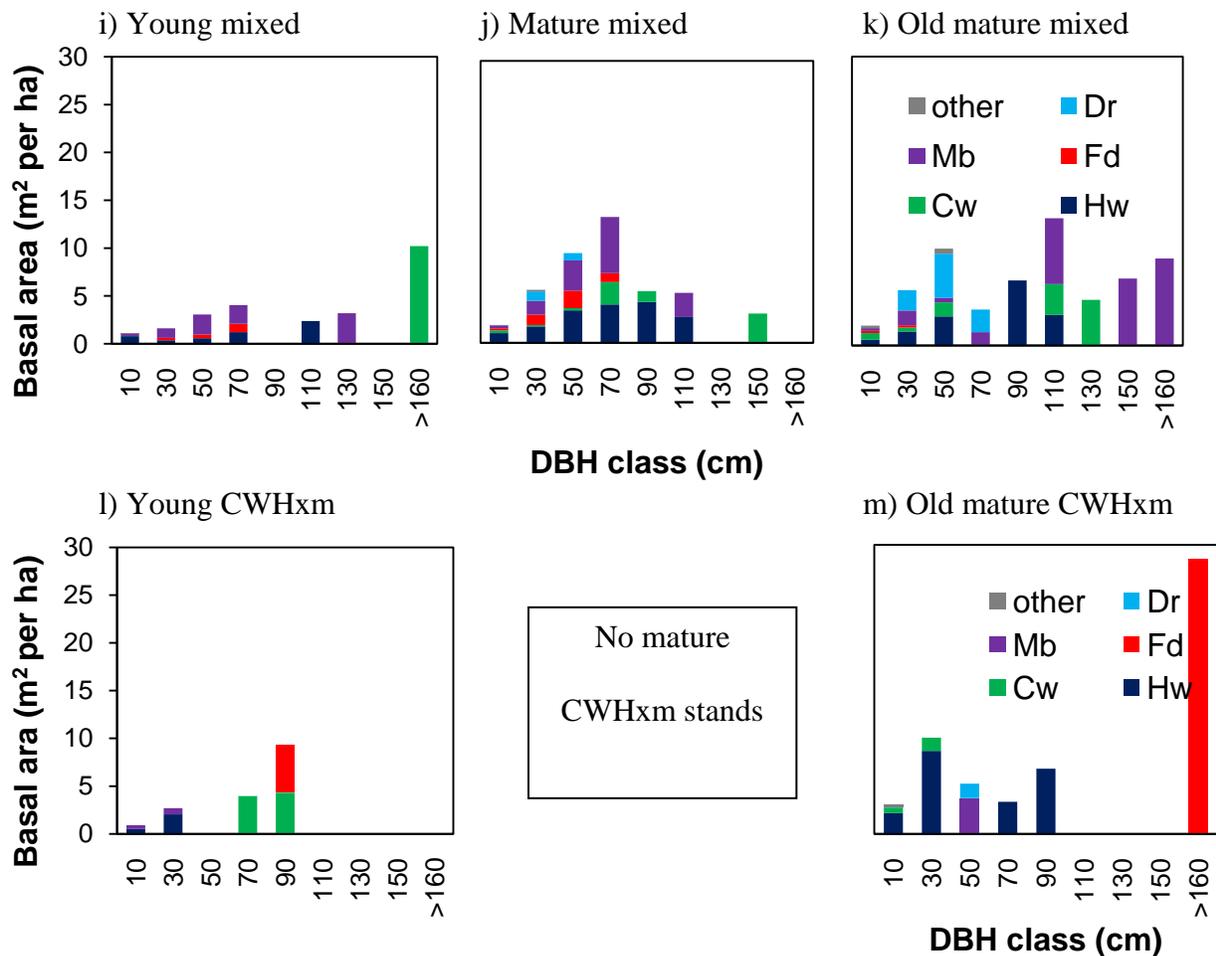


Figure 3-18. cont.

3.5.8 Area of forest cover

The total forest vegetation coverage in the park was found to be 256 ha. This figure does not include areas with trees, native or exotic, in areas with lawns or landscaping. Of the 256 ha most of the forest area was classified into the wet or dry conifer classes (Figs. 3-19 and 3-20). Old-mature forests of all classes covered more area than the two other age classes for a total of 124 ha. Twenty-nine hectares were classified as young forest.

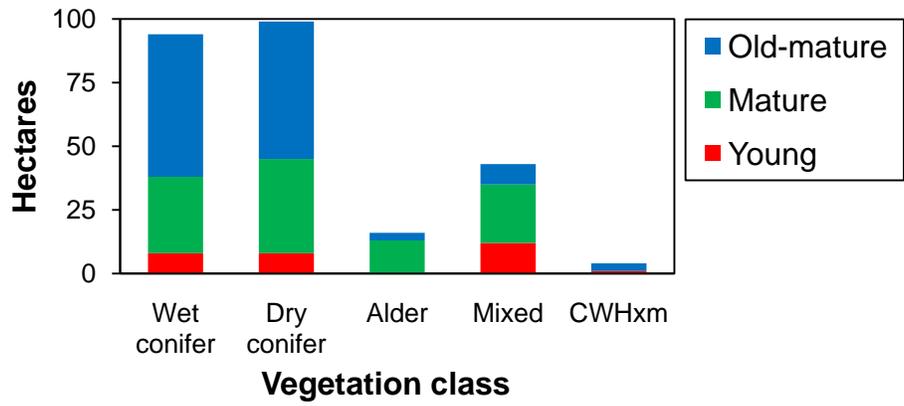


Figure 3-19. Area in hectares of each vegetation and age class.

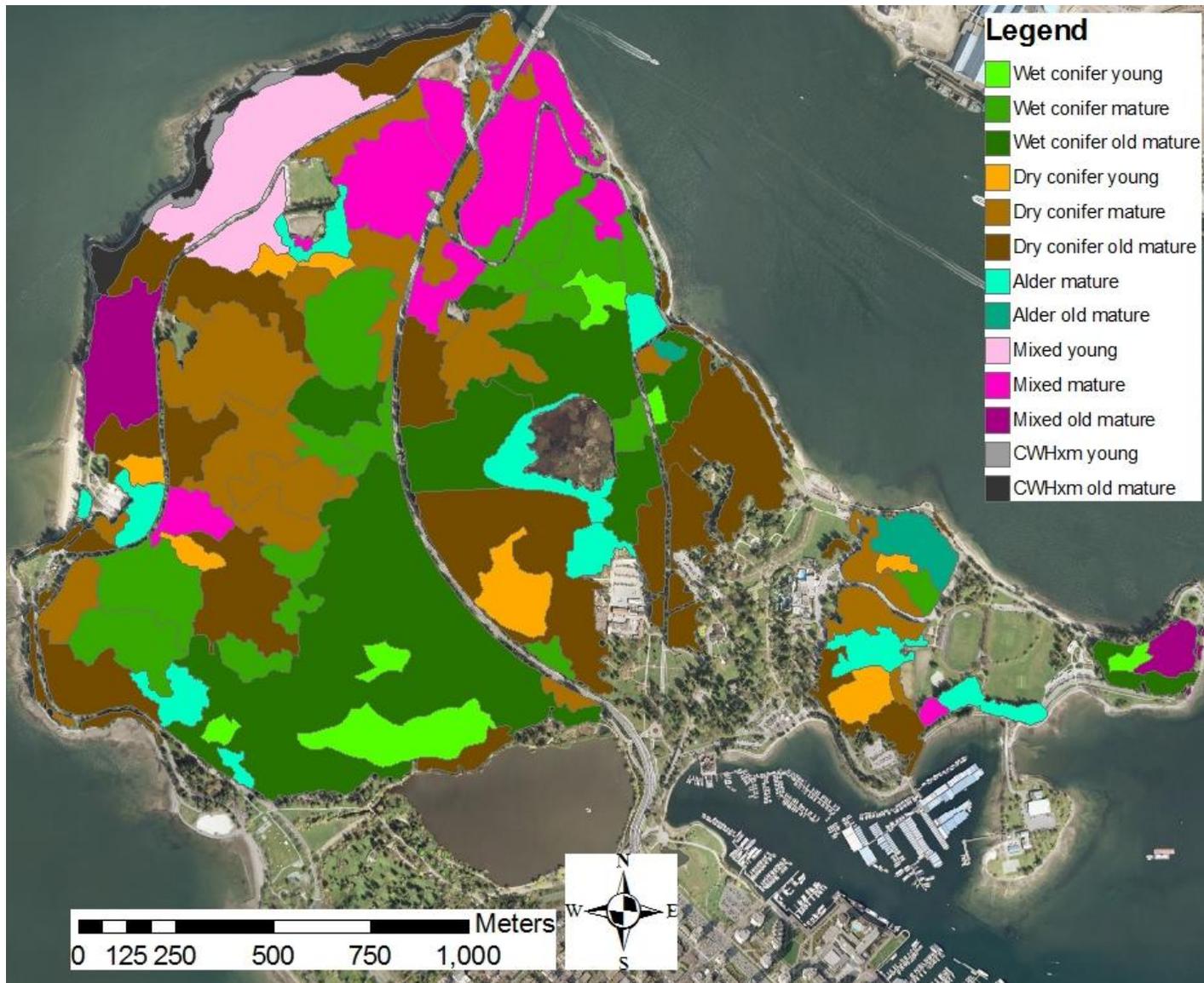


Figure 3-20. Forest vegetation map of Stanley Park overlaid on a 2008 aerial photograph

3.5.7 Canopy cover

Percent light transmission, based on hemispherical photograph analysis for the permanent sample plots, varied greatly between the young forest types and the mature and old mature forest types. Light transmission was less than 25% in the old and old mature forest types, except for the mature alder vegetation type (Fig. 3-21). This vegetation type, as well as the all the young vegetation types have light transmissions greater than 50%. However, the sample size in 5 of the 13 vegetation types with plots was 1.

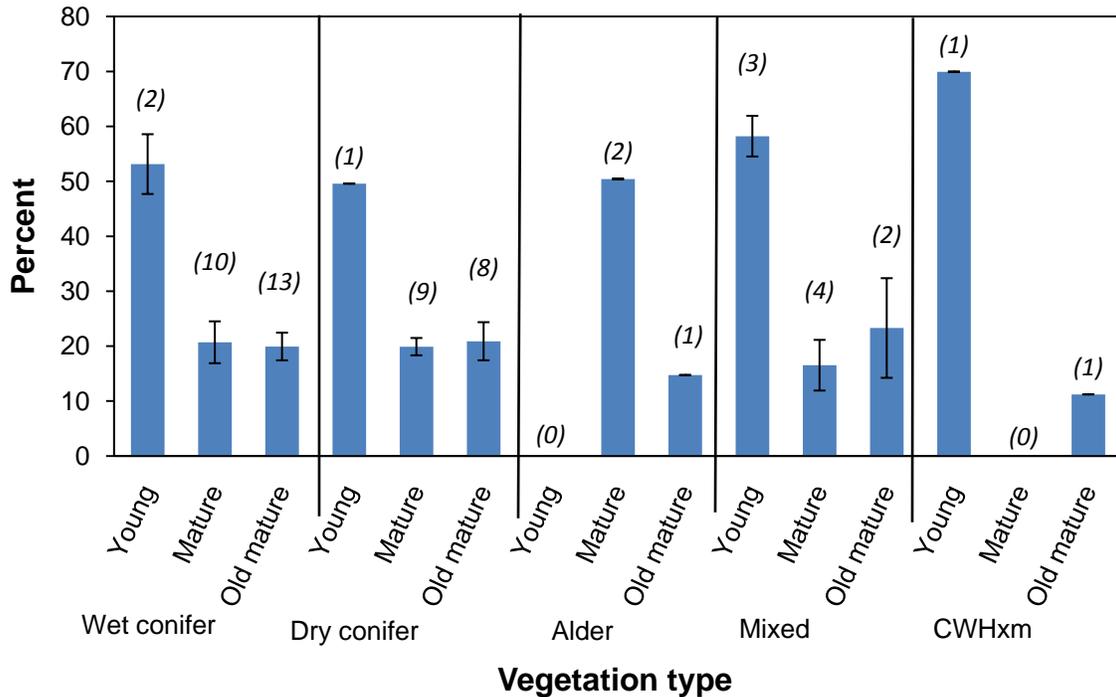


Figure 3-21. Mean light transmission in percent of maximum for each vegetation type. Bars show one standard deviation above and below mean, where applicable. Numbers in parenthesis above each bar indicates number of plots.

3.5.10 Coarse woody debris

Young stands generally contained more hard intact (class 1) coarse woody debris (CWD) than older stands in the same vegetation type (Fig. 3-22). There is a similar amount or more CWD ranging from large partly decayed pieces to soft logs (classes 3 to 5) than class 1 or 2 CWD in most stands. Notable exceptions being the young mixed deciduous/coniferous and young CWHxm vegetation types.

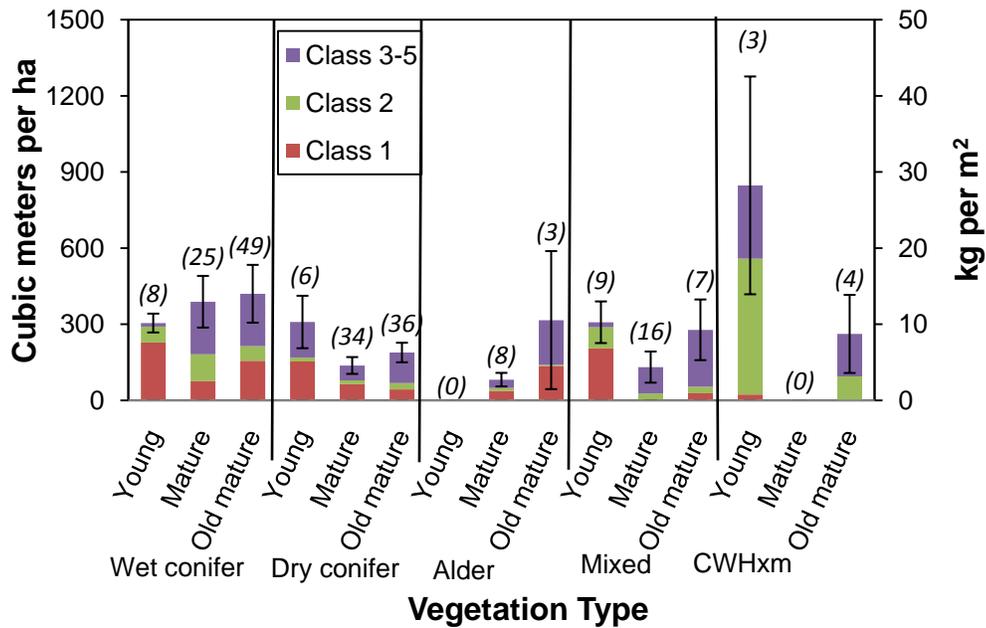


Figure 3-22. Volume per ha and kg per m² of coarse woody debris (> 7.5 cm diameter) by vegetation type. Bars show standard error for the total LWD volume. Numbers in parenthesis above each bar indicates number of plots.

3.5.11 Regeneration

The seedling class was subdivided for analysis into ‘planted’ and ‘natural’ regeneration. The planted group includes only trees planted as part of the restoration project following the 2006 storms. All regeneration data used in analysis come from 2009.

A total of 16 tree species were found in the seedling size class (> 30 to ≤ 130 cm in height). Some of these species were not found in the tree layer in the plots, and these included Norway maple, an exotic species, and Pacific yew and black cottonwood, which are native. Bitter cherry was found it much greater abundance in this class than in the tree layer.

The young wet and dry conifer types have more tree diversity than the older conifer dominated types. Western hemlock was the most abundant species of seedling in 2009 in all vegetation types except the young CWHxm (Fig 3-23). Deciduous species make up most of the individuals of the ‘other’ species present in all vegetation types. No significant difference was found in the number of seedlings per ha in 2009 in the different vegetation and age classes.

Trees of four species (Douglas-fir, grand fir, Sitka spruce and western redcedar) were planted in areas where canopy loss was greatest following the 2006 storm and where planting would not impinge on other objectives such as protecting viewscales. Planted trees were found in 9 of the 13 vegetation types found at Stanley Park. Of these, 4 types had no more than 1 planted species.

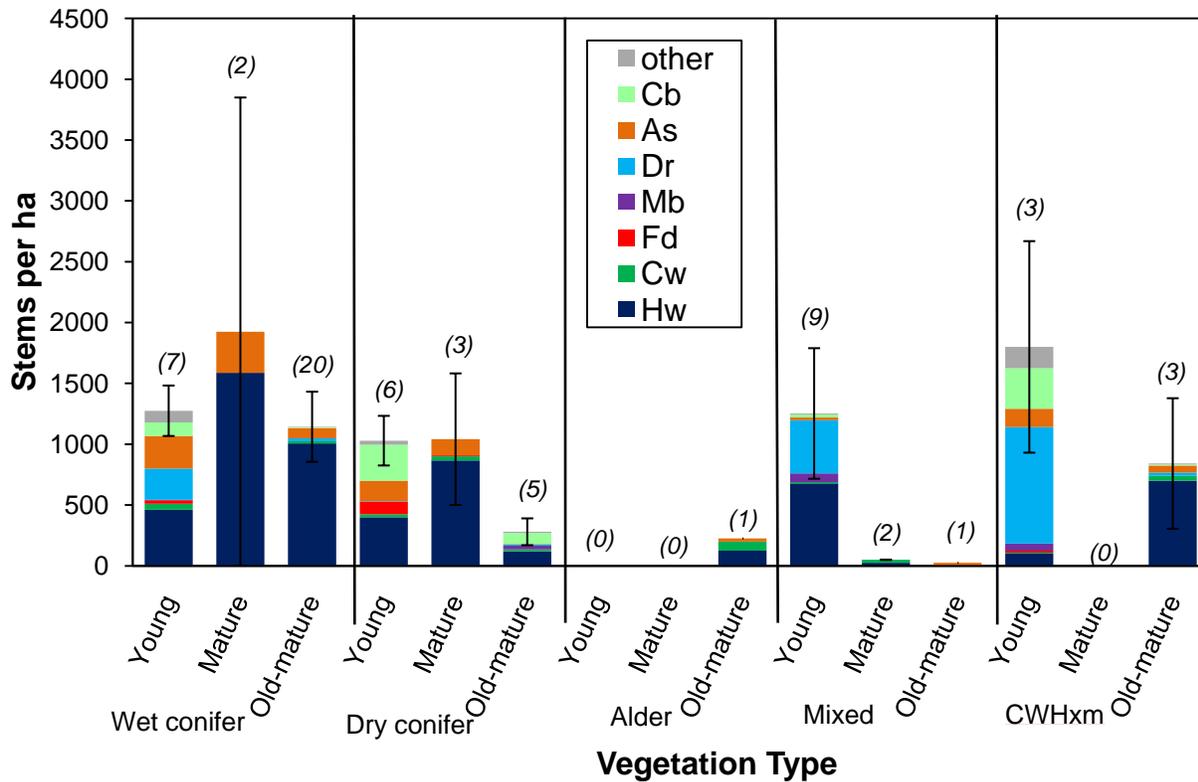


Figure 3-23. Seedling stems per ha by species for each vegetation type. Numbers in parenthesis above each bar indicates number of plots. Bars show standard error for the total seedlings per ha. 'Other' includes arbutus, black cottonwood, chestnut, Norway maple. Cb – bitter cherry, As – Sitka mountain-ash, Dr – red alder, Mb – bigleaf maple, Fd – Douglas-fir, Cw – western redcedar, Hw – western hemlock, other – all other species found in Stanley Park.

Trees of 4 species were planted in windthrow areas based on recommendations by B.A. Blackwell and Associates. Trees were planted in clusters of 2 to 5 trees or individually. Three of the young vegetation types had more than 7 plots with seedlings. Of these three vegetation types the highest density of planting in vegetation types with more than one sample plot was found in the dry conifer vegetation type with 279 planted trees per ha (Fig. 3-24). Sampling was not designed to assess compliance of planters, but to determine the average planted tree density to help project possible future stand conditions. The sample size was small for most vegetation

types so it is difficult draw conclusions from the data, however in all cases planted trees make up less than one quarter the number of naturally occurring seedlings. Total regeneration above 30 cm in height was therefore dominated by western hemlock and deciduous species (Fig. 3-25).

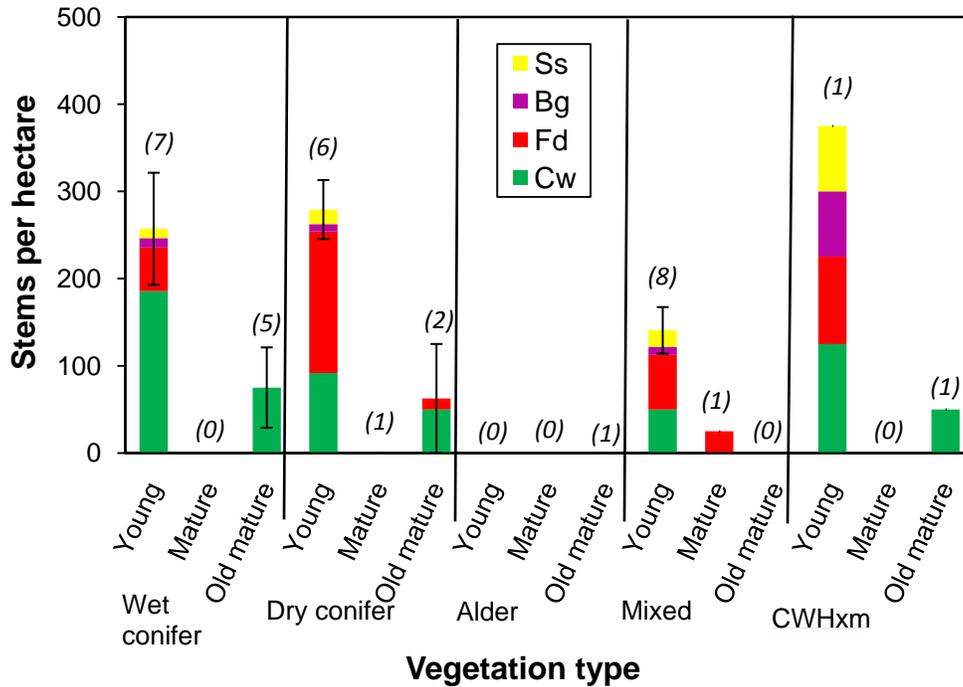


Figure 3-24. Planted stems of each species per hectare in plots within planting blocks by vegetation type. Numbers in parenthesis above each bar indicates number of plots. Bars show standard error for the total planted trees. Ss – Sitka spruce, Bg – grand fir, Fd – Douglas-fir, Cw – western redcedar.

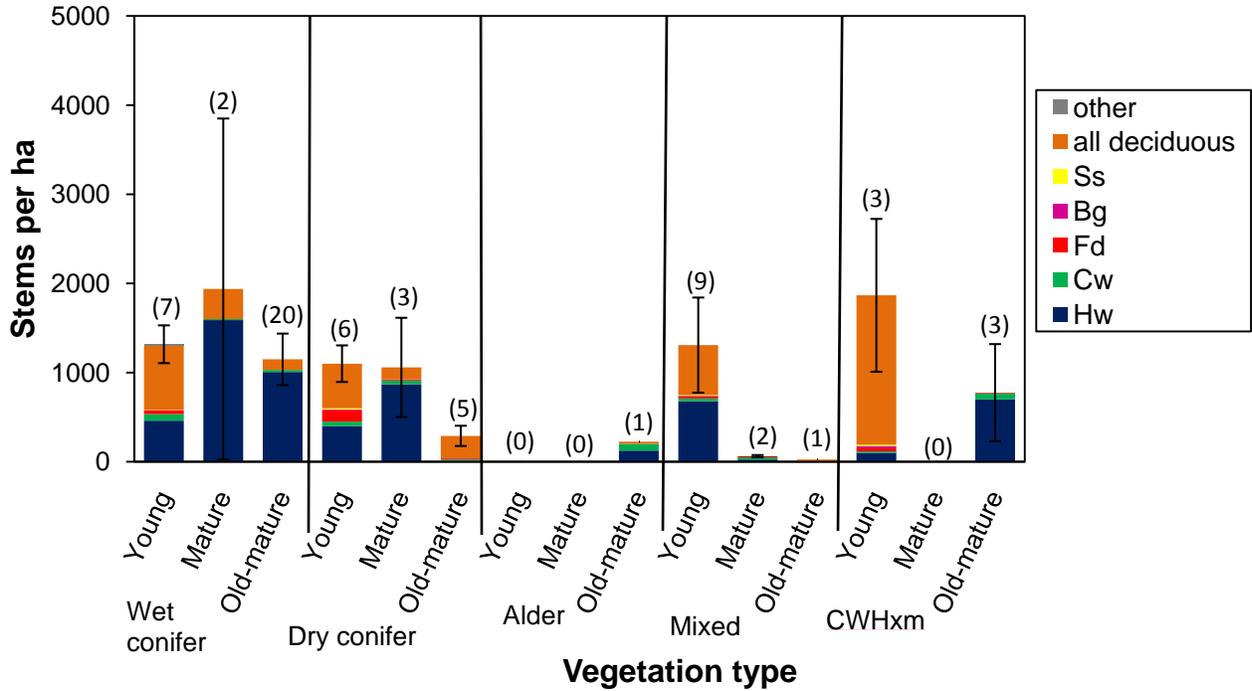


Figure 3-25. Stems per hectare of naturally regenerated seedlings and planted trees combined. Numbers in parenthesis above each bar indicates number of plots. Bars show standard error for the total number of seedlings and planted tree together. Ss – Sitka spruce, Bg – grand fir, Fd – Douglas-fir, Cw – western redcedar, Hw – western hemlock.

The 4 planted species had similar total heights in 2009 (Fig. 3-26). Even though the planted trees had similar heights, fewer than half the grand fir, western redcedar and Douglas-fir were found to be unobstructed by understory vegetation (Fig. 3-27) while 72 percent of Sitka spruce were unobstructed. Douglas-fir grew by an average of over 40 cm between 2009 and the end of the previous growing season, as measured in 2009 (Fig. 3-28). Sitka spruce grew 23.2 cm and grand fir grew 14.8 cm on average in the same period.

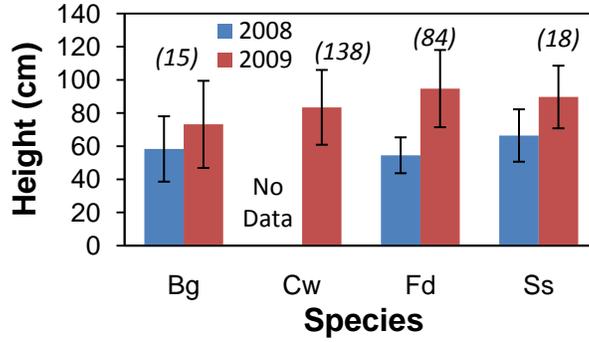


Figure 3-26. Height of planted trees in 2008 and 2009 in cm. Western redcedar have a sustained growth pattern so it was not possible to determine planted tree height in 2008 based on 2009 appearance. Numbers in parenthesis above each bar indicates number of trees. Bg – grand fir, Cw – western redcedar, Fd – Douglas-fir, Ss – Sitka spruce.

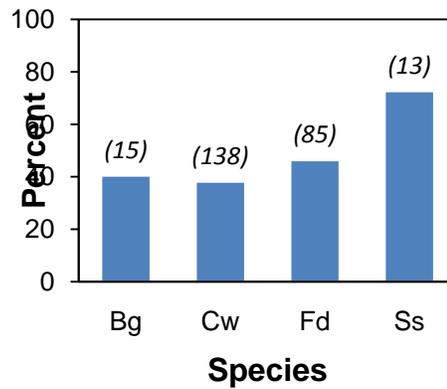


Figure 3-27. Percent of planted trees by species that are unobstructed by understory vegetation. Numbers in parenthesis above each bar indicates number of trees. Bg – grand fir, Cw – western redcedar, Fd – Douglas-fir, Ss – Sitka spruce.

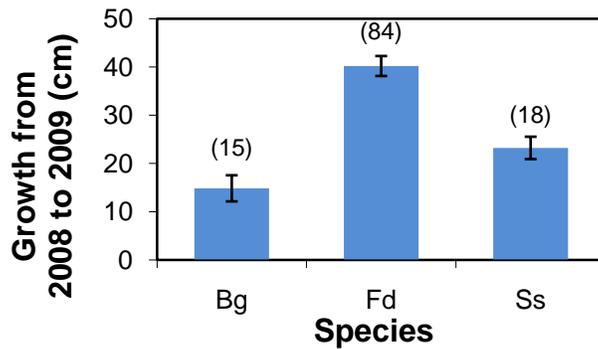


Figure 3-28. Mean growth of planted trees by species across all vegetation types from 2008 to 2009. Bars indicate standard error. Numbers in parenthesis above each bar indicates number of trees. Bg – grand fir, Fd – Douglas-fir, Ss – Sitka spruce.

The germinant size class was divided into 3 subclasses. The subclasses were 0 to ≤ 10 cm, > 10 to ≤ 20 cm and > 20 to ≤ 30 cm. These trees could have established in any year. Most germinants in the plots in windthrown areas sampled in 2009 were found in the smallest size class (Fig. 3-29). Species abundance changed from one size class to another. Western hemlock was an abundant species in all size classes, but there were also substantial numbers of western redcedar in the smaller size classes (Fig. 3-30).

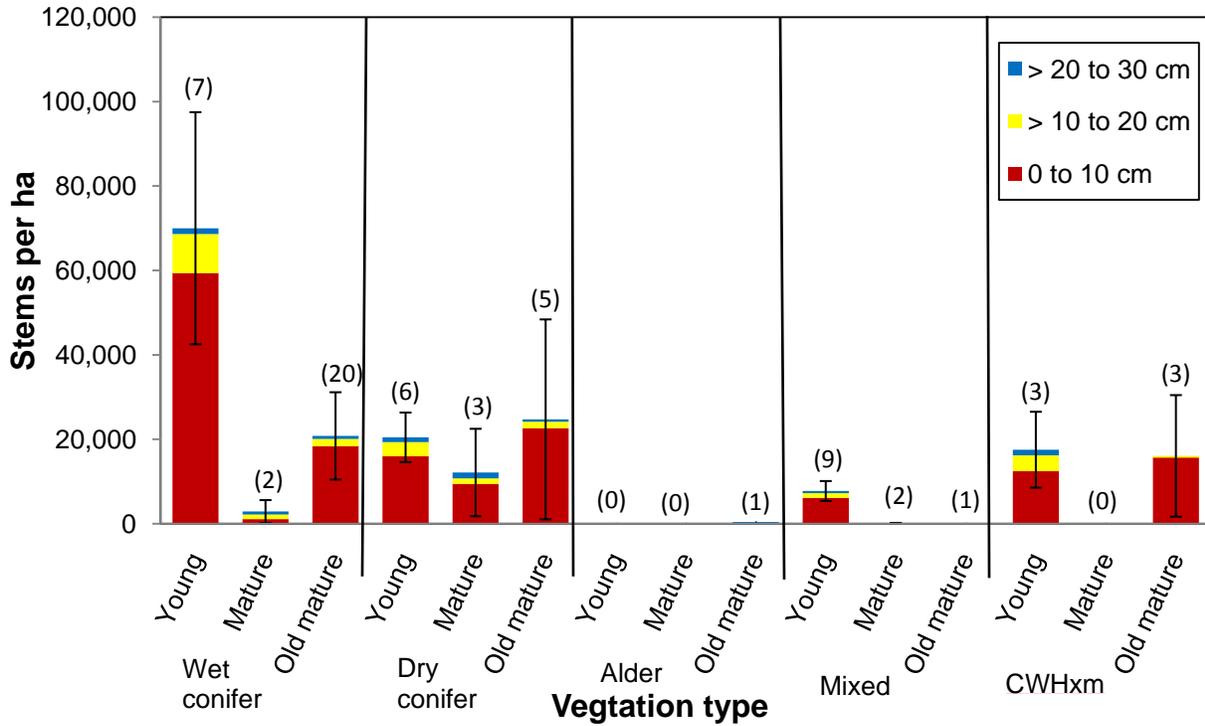


Figure 3-29. Mean number of stems per hectare of germinants in each of three size classes (0 to \leq 10 cm, $>$ 10 to \leq 20 cm and $>$ 20 to \leq 30 cm) for each vegetation type. Numbers in parenthesis above each bar indicates number of plots. Bars show standard error for the total number

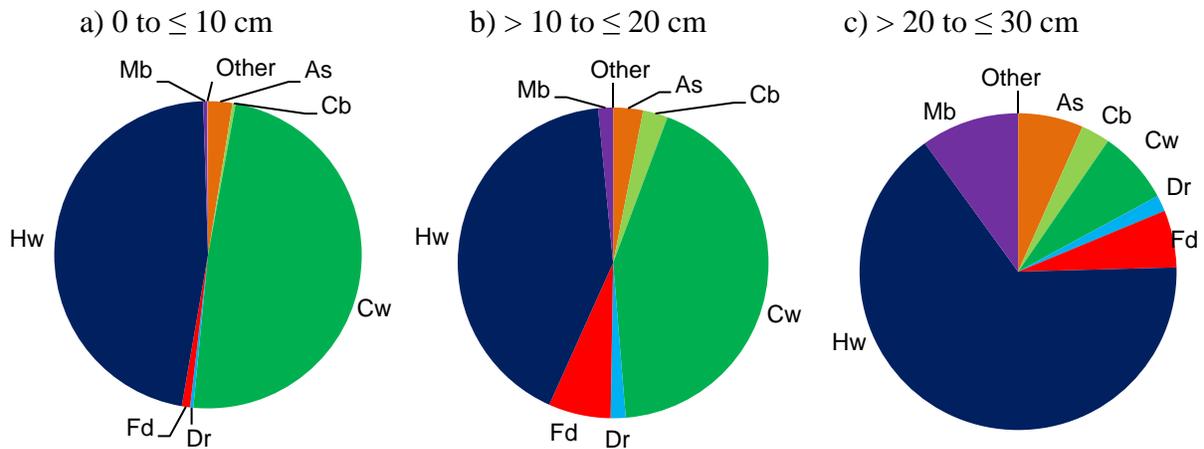


Figure 3-30. Pie charts showing the relative number of each species found in all plots in a) the 0 to \leq 10 cm in height class, b) the $>$ 10 to \leq 20 cm height class and c) the $>$ 20 to \leq 30 cm height class for all vegetation types. 'Other' includes Sitka spruce and white oak.

3.5.12 Understory vegetation

Thirty-two species of understory vegetation covering at least 1% of a plot were found in 2008 and 2009(Appendix 4). It was possible for total percent cover values to be greater than 100% because of layering of vegetation. Only the young mixed deciduous coniferous and the CWHxm vegetation types had less than 100 % total cover of understory vegetation (Fig. 3-31).

The abundance of salmonberry and sword fern (*Polystichum munitum* (Kaulf.) Presl) was particularly important because these are indicator species for wet and dry sites, respectively. Abundance of salmonberry and sword fern generally, though not always, increased from the young to old mature vegetation types. Among the three coniferous vegetation types, salmonberry was most common in the wet conifer vegetation type for each age class. In the young age class, sword fern was most abundant in the CWHxm vegetation type (Fig. 3-31).

Three invasive species of interest to park managers, English ivy (*Hedera helix* L.), holly (*Ilex aquifolium* L.) and Himalayan blackberry, were found throughout the park. Holly was found in more plots than either English ivy or Himalayan blackberry (Fig. 3-32). English ivy also covered more area than the other two species (Fig. 3-33). The CWHxm plots had the lowest incidence of invasive species.

The plots were mapped showing the percent cover of each invasive species (Figs. 3-34, 3-35, and 3-36). Holly was found at low levels throughout the park. English ivy and Himalayan blackberry were generally found near roads or other developed areas. English ivy was

particularly common in plots in north eastern part of the park near the hairpin turn on Stanley Park Drive.

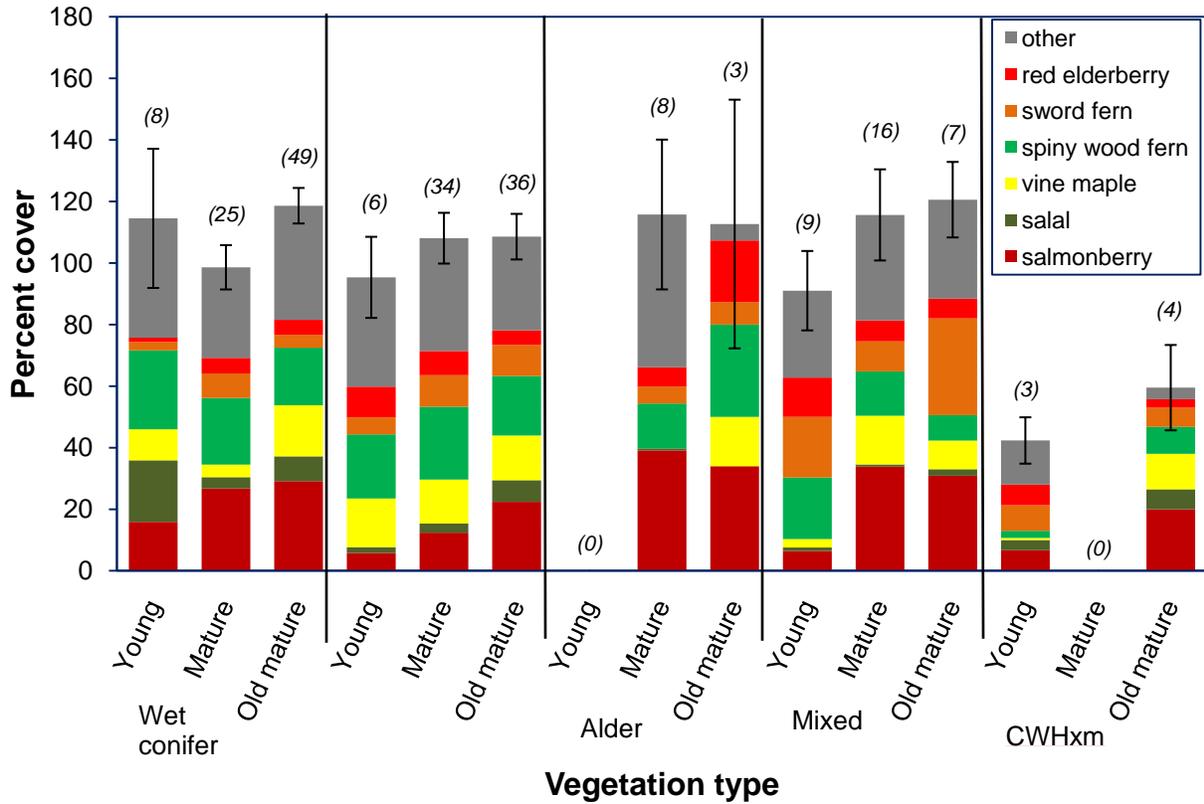


Figure 3-31. Percent cover of understory vegetation in Stanley Park. The 6 most common native species are shown individually. All other species are included in 'other.' Numbers in parenthesis above each bar indicates number of plots. Bars show standard error for the total percent cover of each plot.

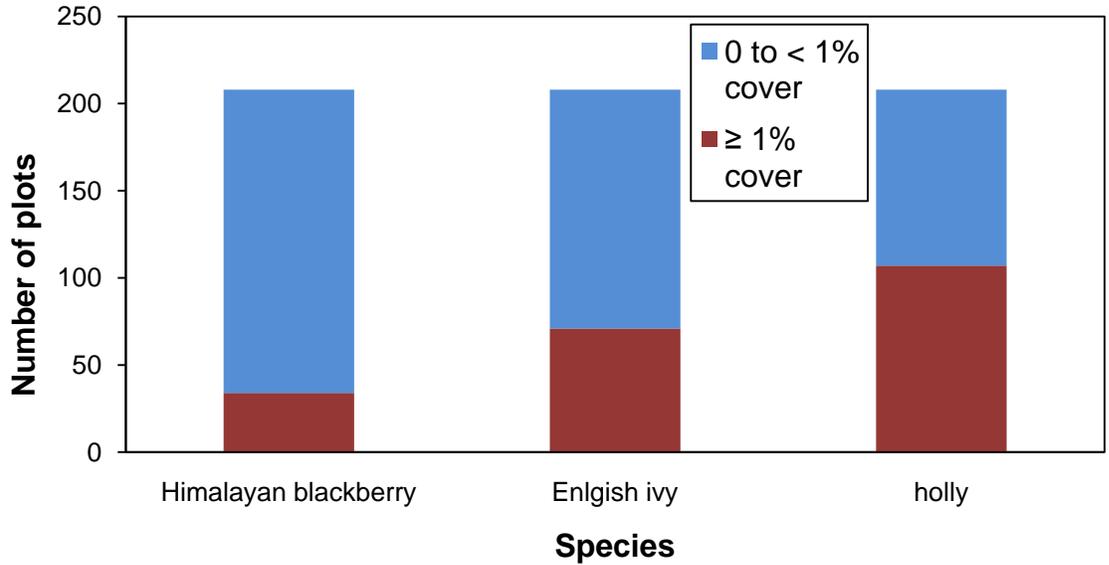


Figure 3-32. Number of plots and with less than or at least 1% cover by each of three invasive species.

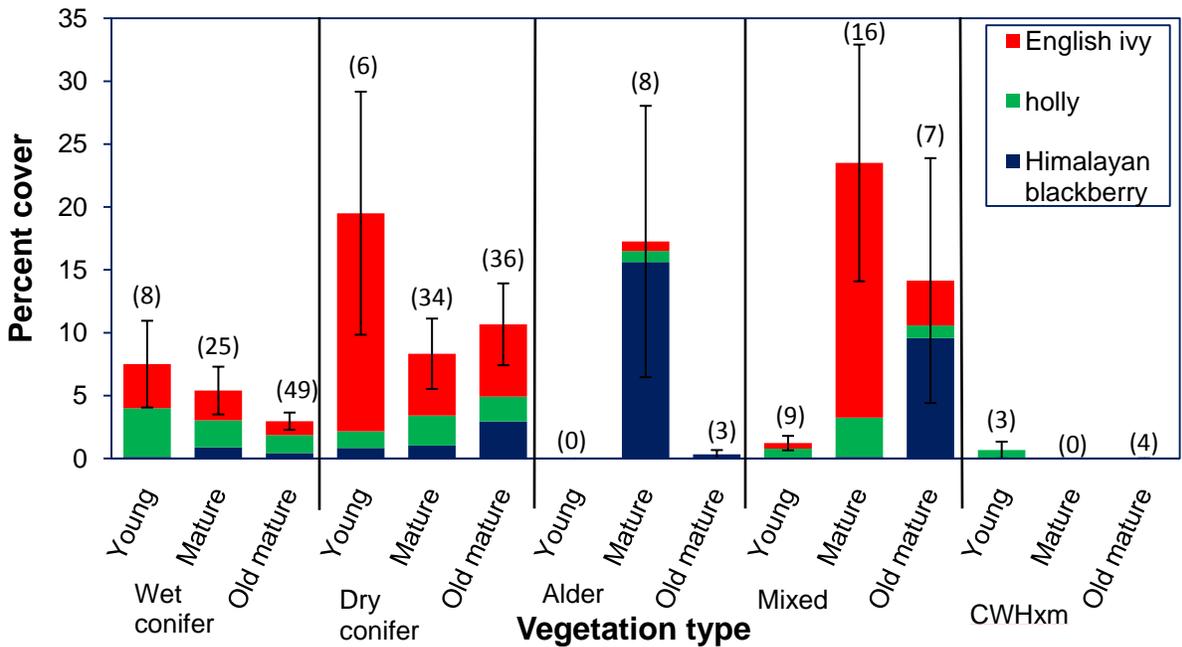


Figure 3-33. Percent cover of three invasive species by vegetation type. Numbers in parenthesis above each bar indicates number of plots. Bars show standard error for the total percent cover of the three invasive species per plot.

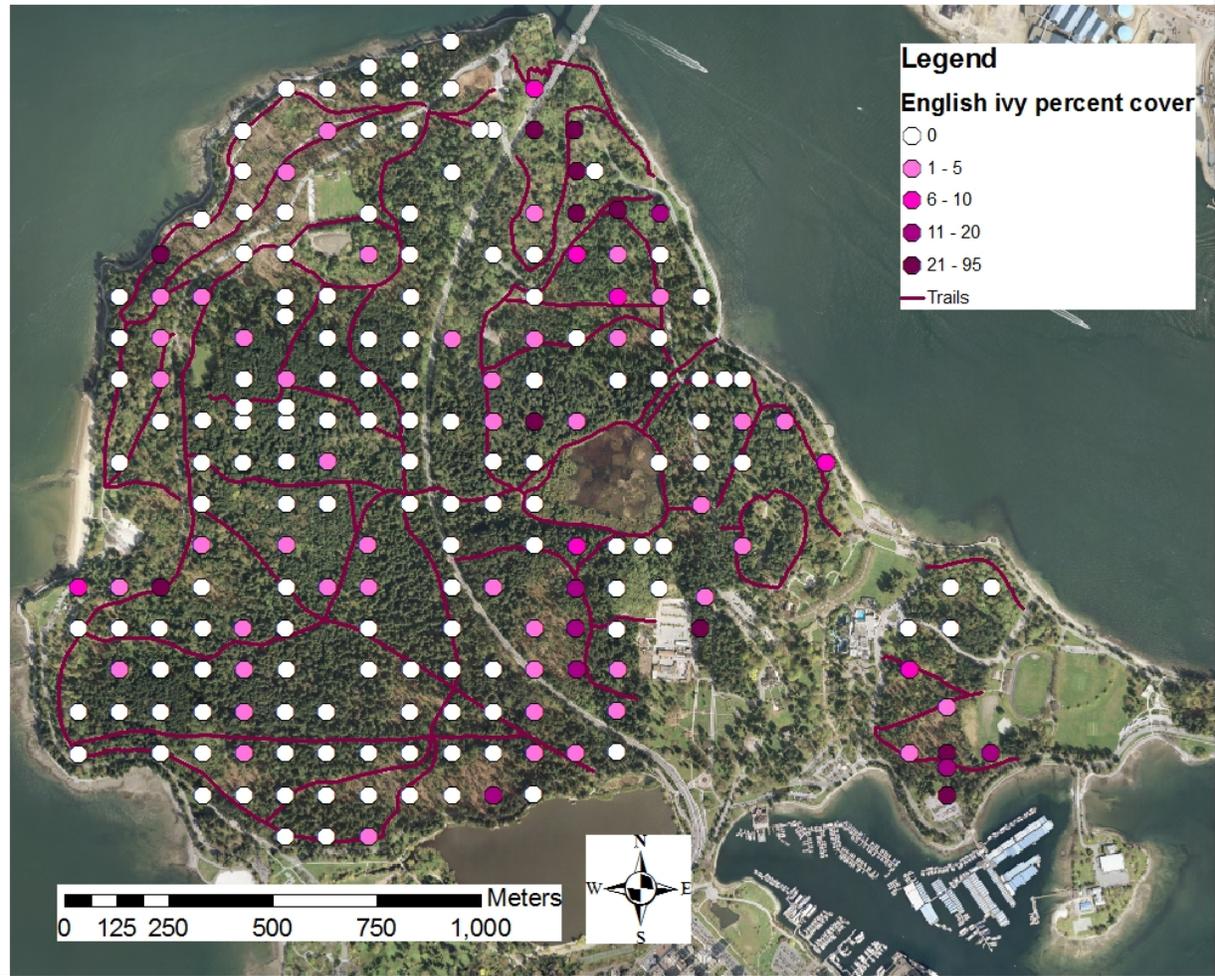


Figure 3-34. Map of Stanley Park showing the percent cover of English ivy at each plot.

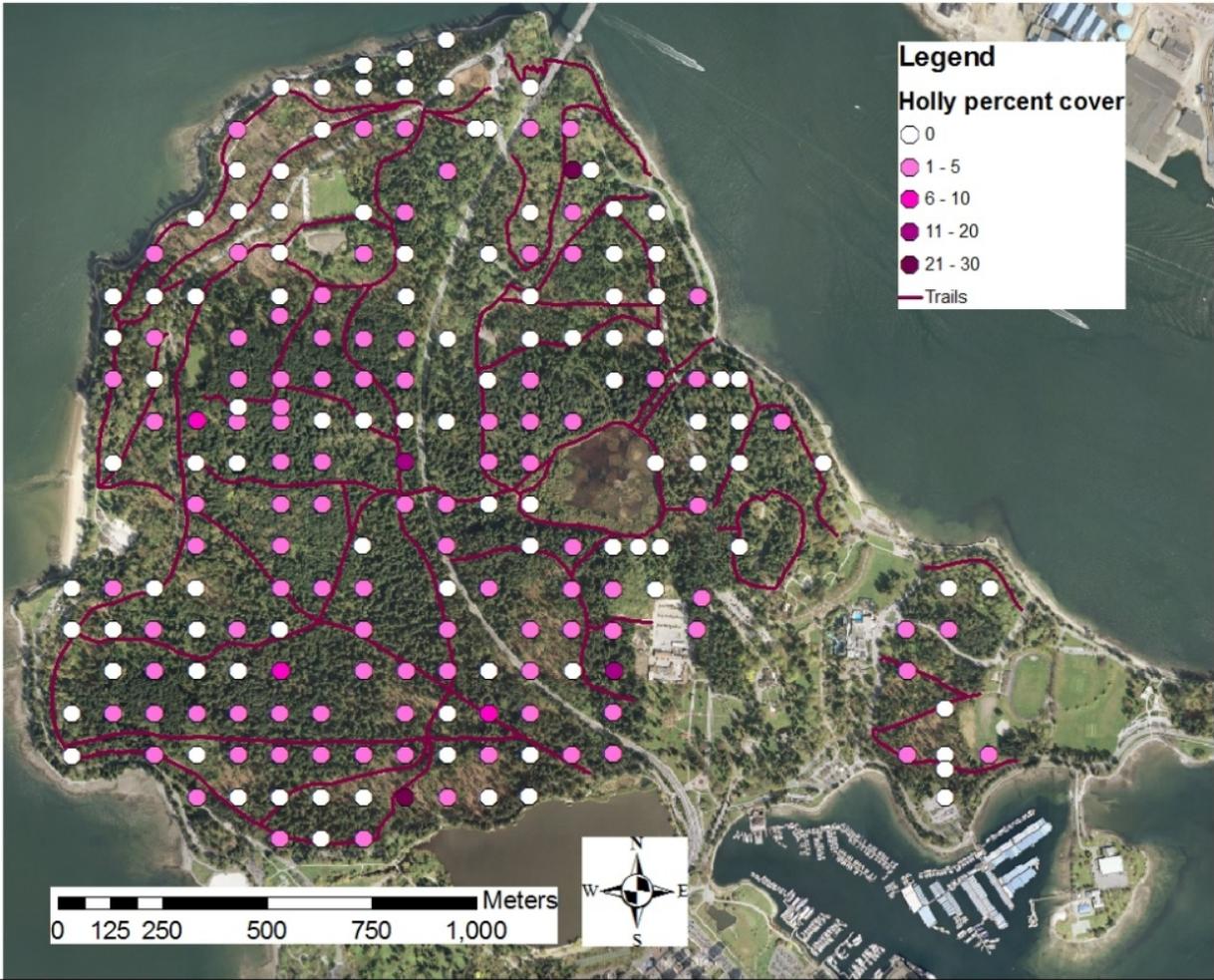


Figure 3-35. Map of Stanley Park showing the percent cover of holly at each plot.

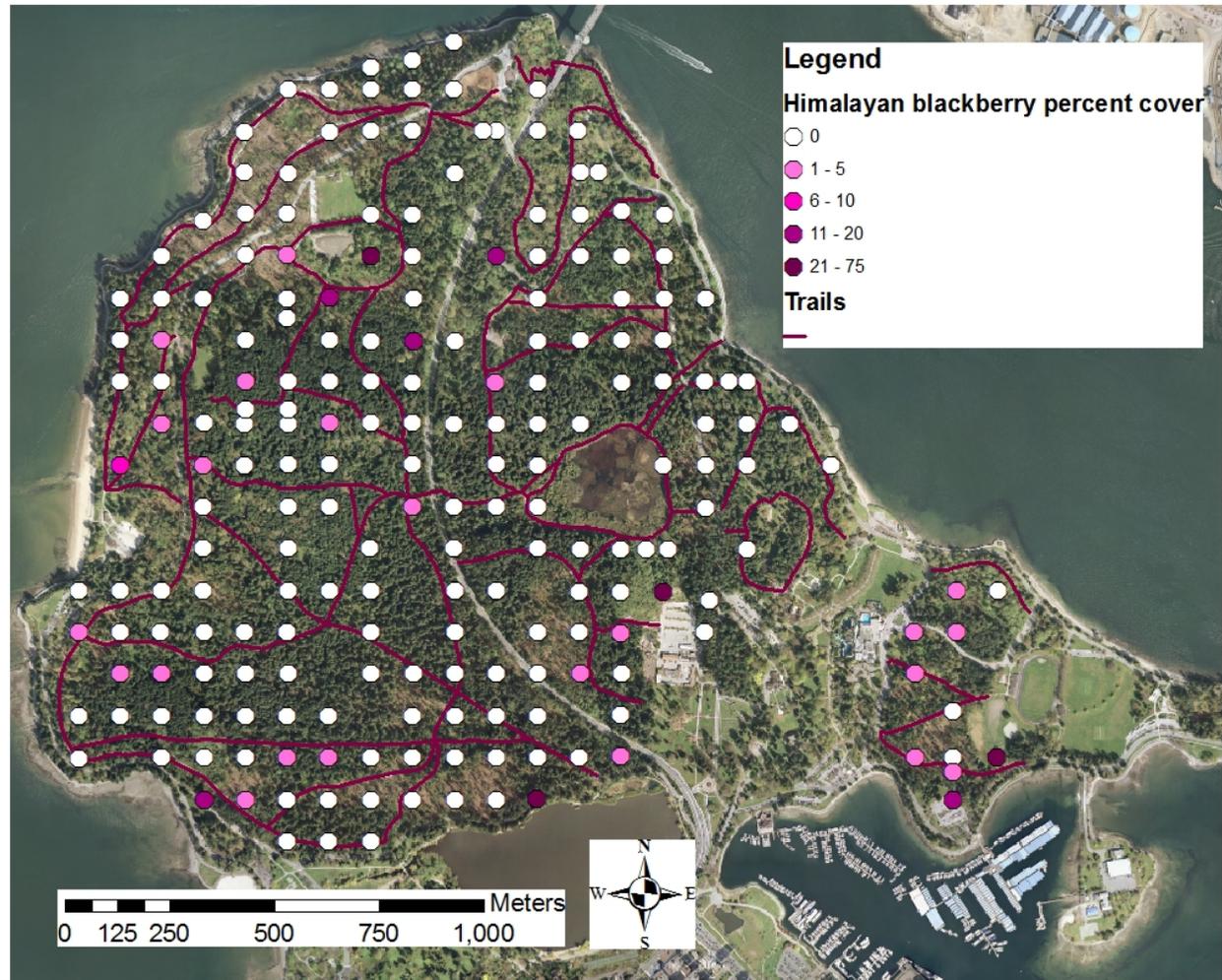


Figure 3-36. Map of Stanley Park showing the percent cover of Himalayan blackberry at each plot.

3.6 Discussion

Windstorms have periodically transformed the forest canopy of Stanley Park. In the two major disturbances covered by the photographic record, the 1962 and 2006 windstorms, large areas of canopy were opened. However, even in the most heavily damaged stands residual trees were present. This is particularly clear following the 2006 storm as the young stand types on average, have basal area per ha values of about one-half those of the older stand types. In

addition to these major disturbances, Kheraj (2007) found 19 windstorms which damaged 10's to 100's of trees from 1900 to 1960 in addition to the much larger storm in 1934. This pattern of low intensity damage punctuated by periodic severe damage has continued to occur since 1960.

Residual trees add vertical heterogeneity to the stand and modify the growing conditions for regeneration. Spies and Franklin (1989) note that Douglas-fir does not regenerate in small gaps. Rose and Muir (1997) found that when the density of remnant canopy trees exceeds 15 stems per ha, Douglas-fir regeneration basal area 70 years after disturbance was less than stands without a remnant canopy. They found that western hemlock tended to replace Douglas-fir as Douglas-fir basal area decreased. Remnant overstory trees compete with regeneration by obstructing light and competing for soil moisture (Harrington 2006).

All stand types contain higher numbers of trees in the 0 to 20 cm DBH class than any other size class. These trees may be of poor form due to low light conditions prior to the 2006 windstorm, and damage incurred during the windstorms in 2006 and the subsequent cleanup operations. These small residual trees maybe a significant source of canopy infill in the disturbed areas. Modelling scenarios presented in the next chapter will explore this possibility.

Red alder regenerated in post-Freda stands are today about 47 years old. The alder in stands identified as old mature alder stands may be older than this. Any trees regenerated after the 1934 storm would be as old as 76. This exceeds the typical maximum age of red alder, which is 70 years, but, some individuals are able to live longer than this (Harrington 1990). In both alder stand types conifers account for most of the trees in the smaller size classes. Over

time conifers are expected to replace red alder in the canopy. Only in the area along the lakeshore west of Beaver Lake is this not expected because of frequent disturbance and a high water table. These stands are seasonally flooded. The aerial photographs show that the stand west of Beaver Lake has been dominated by deciduous trees since at least 1933

In most areas in Stanley Park, regeneration of western hemlock is favoured over time because of low light conditions under the existing canopy. Western hemlock has good seed dispersal and is also able to rapidly colonize larger open areas (Oliver and Larson 1996). Following the 2006 storm, western hemlock was the most common seedling species outside of the driest sites, where red alder dominates. It was also the most common germinant species.

Western hemlock will continue to be the most common species in the Park without management or widespread mortality. Western hemlock regeneration which does not reach the canopy will be capable of surviving in subcanopy layers. Future low intensity disturbances will allow trees in these subcanopy layers to reach the canopy.

Planted tree regeneration was not found to meet the target densities prescribed by B.A. Blackwell and Associates. Of the young wet conifer, young dry conifer, and young mixed coniferous/ deciduous vegetation types, which each contained between 6 and 8 sample plots, no vegetation type contains more than 279 stems per ha. The target for most planting zones was 450 stems per ha. Further sampling is required to confirm that planted tree densities are below targets. Further study may also illuminate the reason for the low densities found.

Planting following disturbance has been effective in changing forest composition. Planting following the 2006 storm included more Sitka spruce and grand fir than is currently present across the park in the tree layer. Most planted Sitka spruce are already unobstructed by understory vegetation. If manual brushing is continued as planned and care is given to protect planted trees from damage by brushing tools, most planted trees are likely to survive and may become overstory trees, provided they are not overtopped by residual trees in mid or upper canopy layers.

There is considerable natural regeneration in the young forest types. While the number of germinants and seedlings in each plot is highly variable, all plots in the young age classes contain both regeneration classes. Understory vegetation may outcompete naturally regenerating trees in some places. Where this occurs, shrubs may greatly slow the growth of trees to the free-to-grow stage. Shade tolerant species such as hemlock and redcedar have a better chance of surviving in these areas.

In areas where individual or small groups of trees were damaged but the canopy remained largely intact, crown expansion of overstory trees, or a combination of crown growth and recruitment from smaller size classes will fill the canopy gaps. In these locations, shade tolerant species, and western hemlock in particular, are most likely to be recruited into the canopy.

Target stand conditions found in the Stanley Park Forest Management Plan (Vancouver Park Board 2009) call for mature and old mature coniferous forest types to contain no more than 30% western hemlock in the canopy. In the mixed deciduous/coniferous stands, all conifer

species together are to make up no more than 30% of the canopy. The overall average density of western hemlock currently exceeds these target stand conditions. However, since western hemlock is disproportionately found in the smaller size classes, the current canopy conditions are closer to the target stand conditions than the average density or basal area suggests. In the young vegetation types western hemlock is the most common species among seedlings and planted trees combined. While deciduous trees and other conifers may overtop some western hemlock, it is likely that in future stands western hemlock abundance will exceed target conditions. The future growth of these stands will be modelled in the next chapter.

Past management appears to have changed the composition in parts of Stanley Park. In particular the Douglas-fir stands west of the Stanley Park causeway are present due to planting after the Freda windstorm. Naturally regenerated Douglas-fir is not found in such uniform stands in unplanted areas in the park.

The major windthrow events in Stanley Park since 1934 have occurred in the fall or winter. The timing of these storms may act to maintain deciduous trees, and in particular bigleaf maple, in the canopy. These trees are generally shedding leaves or leafless at the time of the storms. Because of this, they do not present as large a wind catching area as conifer trees of similar size. In the 2006 storm, conifers appeared to be more likely to be windthrown than bigleaf maple in stands southwest of Prospect Point. Also, the sprouting ability of bigleaf maple allows the species to capture growing space quickly in stands with significant windthrow. Possibly because of this, the stands in the northeast quadrant of the park mapped as mixed coniferous/ deciduous stands appear to have persisted with similar structure since at least 1933.

Understory vegetation cover found in the youngest vegetation types damaged in 2006, was similar to that found in older stands. It likely that understory vegetation cover will increase further in the young stands as the vegetation responds to the high light conditions (Bailey and Tappener 1998). This vegetation is competing with planted and natural revegetation, particularly on the westter sites. Understory vegetation overtops 20 to 40 percent of the planted trees depending on species.

In none of the young stands does the cover of the tallest species of understory vegetation, vine maple, red elderberry (*Vaccinum parvifolium* Smith) and salmonberry, exceed 50% cover. In portions of plots however, salmonberry is the only species present. Brushing around desirable natural regeneration, in addition to planted trees, could be done to hasten the development of a tree canopy, and increase the survival rate of less common tree species.

Himalayan blackberry and English ivy are actively managed in Stanley Park, but holly is not. Workers have been employed by the Park Board to remove Himalayan blackberry seedlings from the recently disturbed areas (Bill Stephen, pers. comm.) while The Stanley Park Ecology Society (SPES) has a volunteer program to remove English ivy from the park (Stanley Park Ecology Society 2010). Although holly is less common than the other two invasive species, it is more widely distributed. Because of this, the impact of holly on park may be underestimated and the control of holly more difficult than the other invasive species.

The total amount of coarse woody debris found in Stanley Park was within the range found by Feller (2003) in other CWH forests. Only the amount of class-two debris in the young CWHxm type exceeded the range found by Feller (2003) of any particular class of CWD in CWH forests. The amount of CWD in the mature and old-mature wet conifer, and young CWHxm vegetation types exceed the maximum load of CWD of 12 kg per m² prescribed in the Stanley Park Forest Management Plan. The amount of CWD prescribed in the plan varies by age of the stand. In some age classes, the prescription calls for less than 12 kg per m². For example in the 90 to 250 age class the prescription is for 3-5 kg CWD per m². This category along with the > 250 year old stand category, which can have up to 12 kg per m² CWD, roughly cover the old mature age class, which is defined as stands in which canopy trees are >100 years old. Some old mature stands are > 250 years old. In these stands some CWD would have to be removed to reach the prescribed level of CWD. Similarly, due to variability in the prescribed range of CWD with stand age, some mature stands likely exceed prescribed CWD levels for their particular age class.

In addition to stand composition, forest managers of Stanley Park are also concerned about the mechanical stability of trees. Given the history of wind damage in the park, the Stanley Park Forest Management Plan aims to create a more resilient forest that will be less likely to be damaged by wind storms, and better able to recover from the effects of disturbance, pollution, and other stresses. Shade from canopy trees reduces both height and radial growth in subcanopy trees (Drever and Lertzman 2001). Mitchell (2003) found that shade reduced radial growth in seedlings more than height growth leading to increased slenderness in seedlings. This result was found even for seedlings exposed to high mechanical stimulus. Stand density has also

been shown to affect slenderness, but excess slenderness can be reversed by thinning high density stands. Mitchell (2000b) found that following thinning retained trees rapidly reduced slenderness, and that the greatest changes were found in the most slender trees. Young regenerating stands are not at risk of windthrow now but may develop into unstable stands without density management.

3.7 Implications for management

The Stanley Park Forest Management Plan sets out a vision of the park's forests as "a resilient coastal forest with a diversity of native tree and other species and habitats that allows park visitors to experience nature in the city". Even in a 'resilient' forest, changes in the forest canopy due to disturbance must be expected. As in other semi-self-maintained forests, changes to the landscape surrounding Stanley Park from coastal forest to urban metropolis and future climate variability may cause other changes in the forest canopy to occur (e.g. Sinton et al. 2000).

Some of these changes may be desirable, others less so. Current and desired forest conditions may also be within or outside the historic range of variability. Landress et al. (1999) note that the range of natural variability is a good starting place for managing a forest. However, the natural range of variability may include conditions which are undesirable or incompatible with uses and values in an urban setting. Even if the natural range of variability is compatible with an urban setting, changes that have occurred due to human activity or other changes in environmental conditions may make this range unattainable. Managers must therefore assess the

goals for each area within the forested portions of the park and determine the compatibility of current and probable future stand conditions with these goals.

In particular, human intervention will be needed to supplement natural processes which have been modified at Stanley Park. The disturbance regime has been modified by the exclusion of fire. This in turn has changed the natural regeneration pathways, favouring species that do not require fire to regenerate. Understory regeneration development has also been modified by the introduction of invasive species. The presence of these species may also change regeneration and disturbance regimes (Fierke and Kauffman 2006). Nutrient cycling pathways likely have been modified by the addition of nutrients from park users and from the removal of woody debris following disturbance.

The process of competition for growing space in Stanley Park has been largely unchanged. Managers may be able to intervene in this process by thinning undesired trees to achieve management goals. Achieving management goals for some of the vegetation types described in this chapter is the topic of the next 2 chapters. The next chapter will explore the development of stands that were initiated following the 1962 windstorm over the next 100 years under various management scenarios.

4. Freda stand modelling

4.1 Introduction

Vegetation modeling was used to determine how different thinning regimes may alter forest development in stands planted following the windstorm in 1962 nicknamed Freda. The results of the modeling exercises will help inform the selection of prescriptions for future management. In particular, the modeling will focus on how thinning could be used to achieve the park managers' goal to create stand conditions less susceptible to windthrow.

Stand stability can be assessed in a variety of ways. Scott and Mitchell (2005) modelled windthrow using tree, neighbourhood and stand level attributes. When predicting tree-level windthrow hazard they found that neighbourhood attributes such as stand density following thinning and canopy gap size were important variables. The model GALES uses a different method to model windthrow of individual trees. GALES uses tree and stand attributes to model the force applied to each tree by wind, and compares that to the modelled resistive moment of each tree (Gardiner et al. 2000a). Trees unable to withstand the force applied by wind are shown as windthrown. Mitchell et al. (2001) modelled windthrow on cutblock edges using stand level data such as stocking, boundary orientation, and time since harvest of the adjacent stand.

Tree slenderness has been considered an indicator of windthrow hazard (Cremer et al 1982; Becquey and Riou-Nivert 1987). Mitchell (2000a) used data from Becquey and Riou-Nivert (1987) to determine moderate and high windthrow hazard thresholds. Cucchi et al. (2005) found that in high density stands trees mutually support each other, which allows slender trees to survive winds that would cause damage in less dense stands of the same slenderness. Even so, as

stands age and density reduces, tree slenderness became a more important factor in determining windthrow probability than stand density.

Stand stability, one of the components of the 'windthrow triangle' (Mitchell 2000a; Fig. 4-1), is the focus of this chapter for three reasons. The first is that managers have the ability to manipulate the attributes of stands to create more or less stable forests (Schelhaas 2008). The second is that stands at Stanley Park are intended to be managed for very long time period. Over the long term, density is not likely to remain sufficient to provide mutual support as stands in Stanley Park. This is because coastal forests commonly experience low intensity disturbance which opens canopy gaps (Franklin et al. 2002). Finally, managers can quickly plot their stands on stand density management diagrams (SDMD) to determine the current status of their stands with respect to the hazard thresholds also plotted on SDMDs in Mitchell (2000a).

Stand hazard integrates several attributes including live crown size, dominance, health, stand structure and tree slenderness (height to diameter ratio) (Mitchell 2000a). The other components of the windthrow triangle are wind exposure and site hazard. Site hazard is found by determining the level of hazard due to soil conditions.

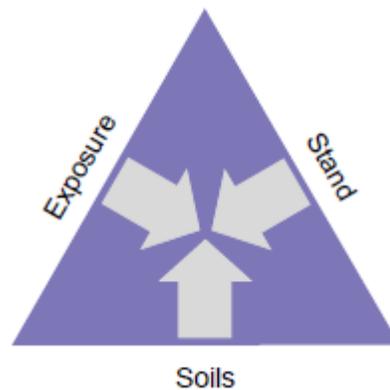


Figure 4-1. The windthrow triangle. Site exposure to wind, stand attributes and soil attributes must be considered together to determine the hazard of windthrow in a stand (from Mitchell 2000a).

Thinning dense stands reduces windthrow hazard in the medium to long term by opening growing space, allowing for more rapid relative diameter increase than height increase (Oliver and Larson 1996). As can be seen in the trajectory of the high hazard threshold in Figure 4-2, as the top height of a stand increases, slenderness must decrease to maintain the same level of stability. The thresholds defined by Mitchell (2000b) were plotted on SDMDs using plot data to determine the top height and quadratic mean diameter of trees at a range of densities for each tree species.

Over time, the slenderness of single canopy stands is reduced as the most slender trees die from competition. This mortality, however, is not sufficient to keep the likelihood of windthrow from increasing. Repeated thinning entries, or frequent low severity disturbances, are required to maintain low to moderate windthrow hazard conditions (Fig 4-2).

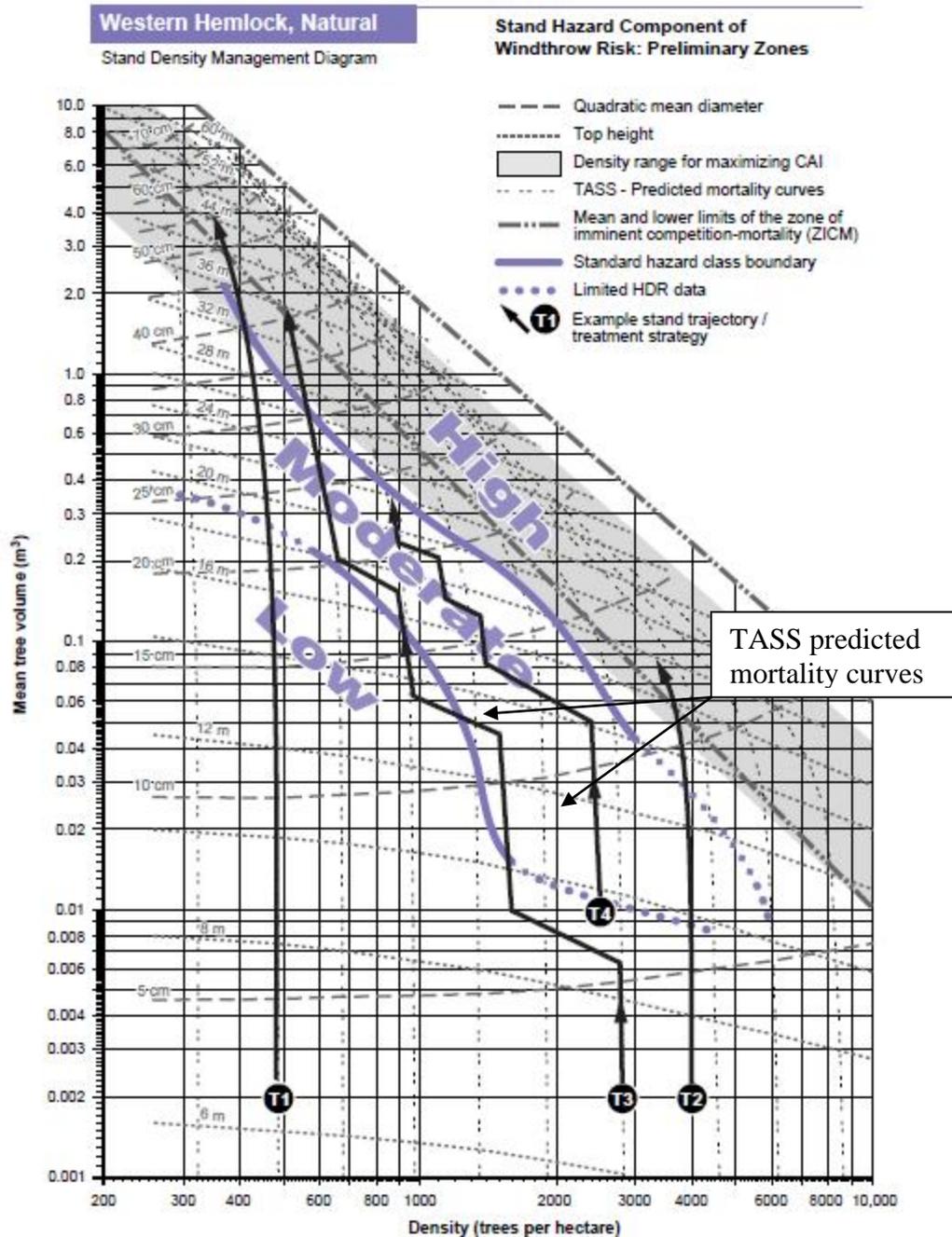


Figure 4-2. Growth trajectories of stands under several management regimes. T1 shows the growth trajectory of stands grown at low density. T2 shows the growth trajectory of stands growth at moderately high density with no thinning. T3 and T4 show the growth trajectories moderate density stands with different thinning treatments applied (From Mitchell 2000a). Only thinning keeps stand from crossing high hazard threshold. Low, moderate, and high windthrow hazard classes separated by purple lines. Note the vertical dotted lines indicating TASS predicted mortality curves.

The Stanley Park Forest Management Plan aims to create a resilient forest that is safe for the public to visit (Vancouver Board of Parks and Recreation 2009). Creating a forest with better long-term windfirmness will address these concerns and help the managers achieve their vision. Accordingly, output from the model was used to assess the potential for future windthrow in Stanley Park. The results from the scenarios were used to determine the effects of thinning on trees slenderness. Thinning was expected to reduce stand slenderness to below high hazard threshold identified by Mitchell (2000a).

4.2 Methods

4.2.1 Model selection

The Pacific Northwest Coast (PN) variant of Forest Vegetation Simulator (FVS) was selected to model plots in Stanley Park. It was selected because it is able to model growth in mixed species forests, it can model a wide range of management interventions, it can be used for the time-scale of interest (100 years), and because appropriate input data were available. The forest types within Stanley Park are similar to those covered by this variant of FVS. It should be noted however, that Stanley Park is outside the geographic range mapped in the PN variant user guide (Forest Vegetation Simulator Staff 2008) (Fig 4-3). The CWHdm subzone corresponds to a portion of the *Tsuga heterophylla* Zone defined by Franklin and Dyrness (1973) (MacKinnon 2003). Coastal portions of the *Tsuga heterophylla* Zone are covered by the PN variant of FVS. Some error in projected metrics may occur because tree growth rates and relative competitiveness of species may differ between Stanley Park and the PN region. However, the general similarity between environmental conditions in Stanley Park and parts of the region

covered by PV variant mean that errors in growth rates and competition effects should be minimal compared to other sources of error.

FVS includes a post-processor called Stand Visualization System (SVS) (McGaughey 2002). SVS displays each stand based on tree-lists produced by FVS. The spatial arrangement of the trees does not represent the actual stand but rather one possible arrangement.



Figure 4-3. Suggested range of use of the PN variant of FVS. National forests are shown in light grey within the suggested PN variant range. Data from these forests was used in the development of the PN variant.

4.2.2 Assumptions of FVS

FVS is a non-spatial, distance independent model which projects individual tree growth (Crookston and Dixon 2005). When projecting tree growth, FVS is unable to take into account neighbourhood-level competition from adjacent trees. Competition is determined from stand

level metrics such as stand density index, crown competition factor and a tree's height rank, among others. Mortality is preferentially applied to shorter trees.

FVS contains two growth models, a large tree model and a small tree model. The large tree model operates by first calculating diameter growth while the small tree model first calculates height growth. The break-point between large and small trees in the PN variant is 7.6 cm (3 in) DBH. The small tree growth model in the PN variant is not considered reliable (Erin Smith-Mateja, Forest Management Service Center - FVS Group, pers. comm). Trees covered by the small tree growth model grow more slowly than expected and are not killed by the mortality models as quickly as expected. For example, shade intolerant species can remain under a dense canopy for decades with little growth in the model. Other assumptions of the PN variant of FVS include no understory vegetation, no environmental change and no regeneration aside from sprouting of a few species following thinning (Crookston and Dixon 2005). Regeneration however, can be added by the user.

4.2.3 Calculation of stand level statistics

FVS was used to calculate stand level statistics including density, basal area, top height, stand density index and height to diameter ratio. FVS was used to convert the US customary units for density and basal area into metric units. Other conversions were done in MS Excel.

The Society of American Foresters (2008) describes stand density index (SDI) as a measure of relative stand density based on the relationship of stem density and quadratic mean diameter. SDI can be calculated based on Reineke (1933) as follows:

$$SDI = N \left(\frac{D_g}{10} \right)^{1.605}$$

where: D_g is the quadratic mean diameter of trees in the stand, and

N is the number trees.

Curtis and Marshall (2000) show the calculation for quadratic mean diameter (QMD), also known as the average diameter by basal area, is the calculated as follows:

$$QMD = \sqrt{\frac{\sum_{i=0}^n x_i^2}{n}}$$

where: x_i is the diameter of the i -th tree and,

n is the number trees.

4.2.4 Modelling unit

FVS can project the growth of trees in tree-lists supplied by the user. In this study, tree-list data came from field measurements of 11.28 m radius permanent or temporary sample plots (described in Chapter 3, section 4.2, above). The plots used in this study are expected to adequately represent local stand structure because they are relatively large compared to the spacing between overstory trees.

Five plots were selected for modelling from stands initiated after the windstorm Freda. The plots labelled ‘dry’ are from the mature dry conifer vegetation type, the plots labelled ‘wet’ are from the mature wet conifer vegetation type.

In the plots selected, well-expressed dominant individuals are rare because of the dense and regularly spaced planting regime and the uniformity of the planting sites. The establishment regime included the removal of downed trees, piling and burning of debris, and planting of Douglas-fir at up to 2500 stems per ha (Bill Stephen, pers. comm.). One of the five plots included a veteran tree which survived the 1962 storm. The other plots include only trees that regenerated following the storm. Plots with a single canopy were selected preferentially model the most homogenous post-Freda stands (Table 4-1; Fig. 4-4 a-e). Even so, one of the modelled plots included sub-canopy trees.

Plot data entered into FVS were multiplied by a factor which allows output to be generated in units per acre. Since the sample plots were 400 m², the factor was 10.124. In FVS, each tree record is initially multiplied by this factor. A plot is taken, therefore, to encompass the entire range of initial variability found in a 1 acre area. In this case each plot selected is taken to represent a stand, which covers an area larger than 1 acre.

Where multiple plots were sampled within one stand polygon, data from only one plot was used in modelling. Data were not combined from two or more plots because, when given data from multiple plots, FVS merges the data together (Dixon 2002). The resulting stand appears as the average of the input plots. Depending on the composition of each plot the resulting model stand may not adequately represent the actual stand. For example, if the two plots contain a single layer canopy but differ in height, the resulting model stand will appear to be multi-layered.

Table 4-1. Plot attributes in 2008.

Plot name and treatment	N	Ht_{top}	QMD	D_{max}	BA_{tot}	BA_{species}	SDI	HDR_{mean}*	HDR_{range}*	SS	Plot
Dry low density	375	29.1	39.8	64.8	46.7	Fd – 100%	312	71	45 – 114	5	225
Dry moderate density	500	28.6	27.7	47.0	30.3	Fd – 98% Hw – 2%	234	96	69 – 184	5	333
Dry high density	625	30.8	32.3	52.8	50.8	Fd – 98% Cw – 2%	369	79	67 – 128	5	341
Wet moderate density	450	28.3	38.9	68.1	53.5	Fd – 94% Hw – 6%	361	74	19 – 108	7	226
Wet high density	500	31.1	33.9**	64.2** 289.3***	207.3	Cw – 79% Fd – 14% Mb – 6%	1093	72**	48 – 88**	7	330
* Trees over 2.5 cm DBH only			** Fd and Mb only		*** Cw						

Key to attributes:

N – Trees per ha – residual trees only (N per ha)

BA_{species} – Percent basal area by species

Ht_{top} – Top height (m)

SDI – Stand density index

QMD – Quadratic mean diameter (cm)

HDR_{mean} – Mean height to diameter ratio

D_{max} – Maximum diameter (cm)

HDR_{range} – Range of height to diameter ratios

BA_{tot} – Total basal area (m² per ha)

SS – Site series in the Coastal Western Hemlock Dry Maritime subzone

a) dry low density



b) dry moderate density



c) dry high density



d) wet moderate density



e) wet high density



Figure 4-4 a-e. Photographs showing views from the centre of plots used in Freda stand modelling.

4.2.5 Data Transformations

Tree level data for height and diameter were collected in meters and centimetres, respectively. SAS Version 9.2 (SAS Institute 2008) was used to convert these data into feet and inches, respectively, as required by FVS. SAS was also used to calculate percent live crown from total height and height to base of live crown measurements. The data were then entered into a Microsoft Access database which is the input data file type used by FVS. FVS output was then converted to metric units for presentation.

Site data were also entered into an Access database with the following conversions. As required by FVS, plot size was entered as the number of plots required to equal 1 acre which was 10.124. FVS determines site index using plant associations. By comparing species lists for each site series with the plant associations listed for the PN variant, the plant association which best corresponds to each site series was selected (Table 4-2).

Table 4-2. Forest Vegetation Simulator plant association code input for each Coastal Western Hemlock Dry Maritime site series.

CHWdm	FVS PN Variant plant association
01	40 = western hemlock/ salal-evergreen huckleberry
05	43 = western hemlock/ salal/sword fern
06	62 = western hemlock/ evergreen huckleberry (coast)
07	30 = western hemlock/ sword fern-foamflower
12	34 = western hemlock/ skunk cabbage

Other site data were generalized. The ground elevation ranges from 1 to 71 m (3 to 233 ft) above sea level at Stanley Park. For modelling, however, the elevation was entered as 100 feet (30 m) above sea level for all plots. Aspect for all plot surveyed in 2008 was recorded as

one of 8 possible cardinal or ordinal directions. Data collected in 2009 included the aspect bearing. These values were converted to the nearest cardinal or ordinal direction before entry into the Access database. FVS also requires that the U.S. Forest Service region and National Forest closest to the plot location be entered. In this case the region was '6' (Pacific Northwest) and the forest '9' (Olympic National Forest). This location setting adjusts the growth equations used by FVS.

Sitka mountain-ash was removed from the data for modeling in FVS. This was done because Sitka mountain-ash is not supported by FVS and does not reach canopy size in this region. This species accounted for 4.6% of all measured trees greater than 1.3 cm tall.

4.2.6 Identification of canopy layers

Canopy layers were identified to differentiate overstory conifer trees from overstory deciduous trees, and from trees of both types in lower canopy layers. Once trees were assigned to a canopy layer, FVS could be used to calculate average height/diameter ratio and other metrics by layer. In these scenarios the default settings for maximum stand density index (maxSDI) and height growth were used. MaxSDI is the highest value of SDI allowed by FVS. The default setting is 950 for all species found in the PN variant (Forest Vegetation Simulator Staff 2008).

Several plots were also modelled using the plant association specific maxSDI. No difference in canopy stratification was found between the runs using the default maxSDI and the plant association specific maxSDI. Also, the standard time step of 10 years for a simulation period of 100 years was used.

Layers were identified by viewing the Stand Visualization System displays of each plot for each time step. Groups of trees which followed the same growth trajectories over the modeling period in the visualizations were assigned to a layer. For example trees that were initially in the subcanopy which reached the canopy by the end of the simulation were assigned to a canopy layer different from both trees that were part of the canopy over the entire modeling period and subcanopy trees which failed to reach the canopy by the end of the modeling period (Fig. 4-5 and 4-6). Coniferous and deciduous trees were not placed in the same canopy layer since trends in slenderness were plotted by canopy layer and compared to thresholds for windthrow hazard. The thresholds have been published only for coniferous species. FVS allows only up to 3 tree value categories. These categories were used identify each canopy layer in a plot in the database. For stands with more than three canopy layers the tallest two coniferous layers were differentiated from each other and all other layers. The other layers will grouped into one category but no analyses were done on this category.

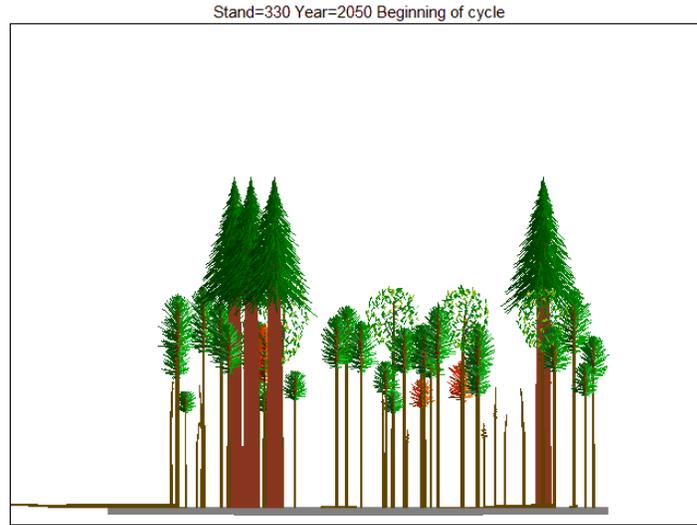


Figure 4-5. Stand Visualization System profile for a plot with 3 layers, a veteran overstory layer of western redcedar, an overstory layer of Douglas-fir and an overstory layer of bigleaf maple. Trees that died during the previous 10 year time step are shown with red crowns. Trees without crowns are standing dead trees. See Appendix 1 for a key to the shapes of trees by species.

a) 2060

b) 2080

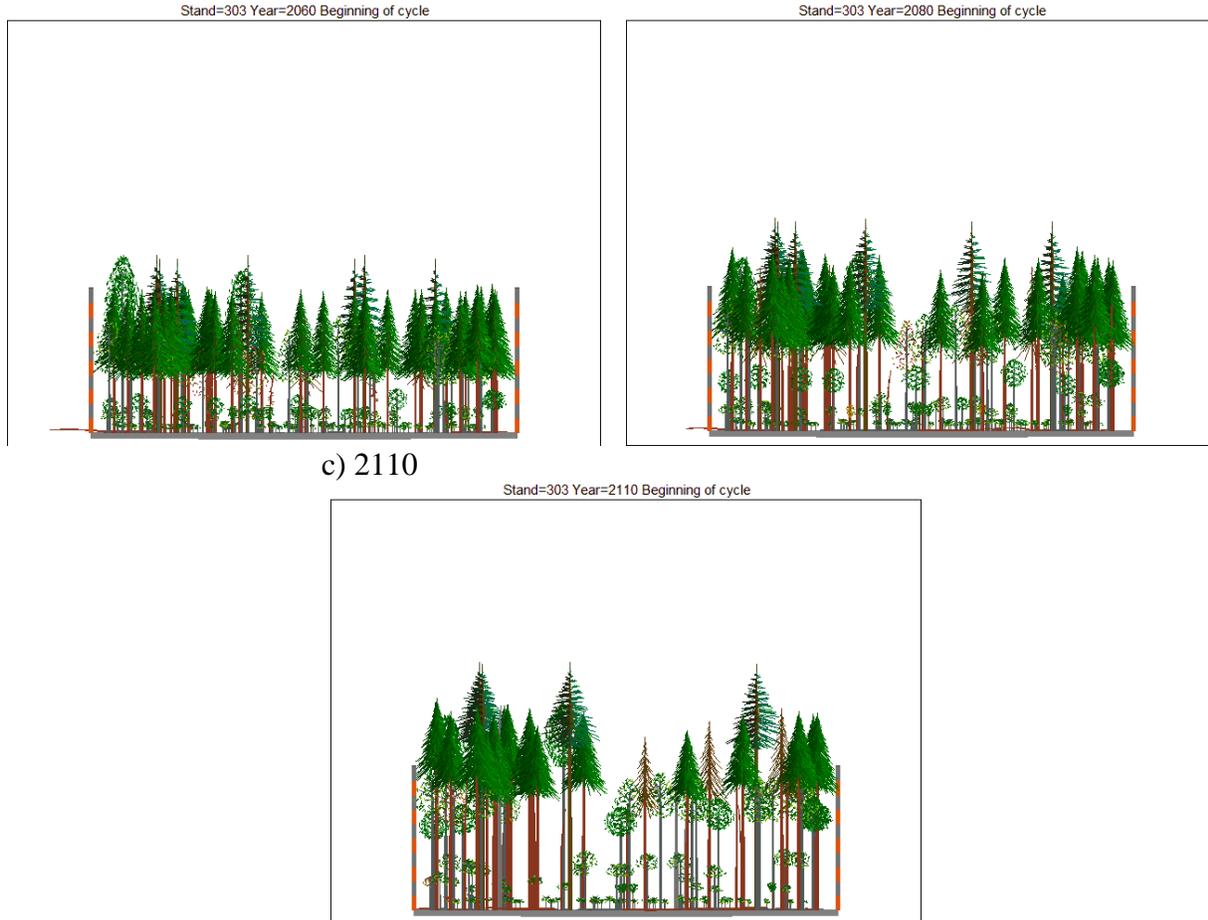


Figure 4-6. This plot appears to develop into 5 layers over the period 2060 to 2110. There is an overstory conifer layer, an overstory deciduous layer which is not growing as fast as the conifers, and 3 subcanopy deciduous layers, each on slightly different growth trajectories. These understory deciduous layers are composed of red alder, paper birch, and bitter cherry. This development pattern is not typical of coastal western hemlock forests because deciduous trees are not usually retained under a coniferous canopy. See Appendix 1 for a key to the shapes of trees by species.

4.2.7 Treatment scenarios

Three scenarios were modeled to explore the effects of silvicultural interventions. Two scenarios included thinning; 1 scenario was a no treatment option. The first thinning scenario was based the prescription prepared by a consulting forester, Craig Farnden, R.P.F., and found in

Appendix 10 of the Stanley Park Forest Management Plan (Vancouver Park Board 2009). In the other thinning scenario more trees were removed in each entry to provide contrast. The no treatment scenario shows stand development without management intervention. The results of these scenarios provide examples of a range of management possibilities, which will help Stanley Park forest managers determine what level of thinning best achieves their management goals.

The prescription in the Stanley Park Forest Management Plan reduces the density of Freda stands to between 200 and 250 stems/ha with thinning entries at 5 year intervals in which no more than 25% of stems are removed in each entry (herein, the ‘planned thinning’). This prescription recommends thinning from below, and also recommends that damaged or diseased trees be removed as well as trees that interfere with the ability of workers to fell trees and equipment used to remove trees. This prescription was modeled as thinning from below to a residual density of 75% of initial density. In FVS, thinning from below removes trees with the smallest diameters until the target residual density is reached. In this scenario, only trees > 15 cm (6 in) were considered for removal since the focus of this prescription is increasing growing space for existing dominant trees. The final density targets only applied to the trees > 15 cm. The scenario was applied to all species except western redcedar. All sizes of western redcedar were retained as this species is recommended for planting in other areas of the park. Retaining these trees also better fit the vision of the forest management plan. The number of thinning entries was either 2 or 3 depending on the initial stand density.

The scenario modelled departs from the prescription prepared by Craig Farnden, R.P.F., in the following ways:

- 1) seriously damaged or diseased trees were not removed preferentially, since they could not be identified in the model,
- 2) trees were not removed to facilitate removal of subsequent trees or to create space for equipment,
- 3) health, vigour and form of the trees were not taken into account since these are not characterized in the model, and
- 4) no preference was given to remove trees near the largest trees, since the model is non-spatial.

Because of these differences, the modelling results provide only an approximation of the changes that would take place in the stands that each plot represents under the planned thinning prescription.

The second thinning scenario was similar to the first, except that 40% of stems were removed in each entry (hereafter, the 'heavy thinning'). The same number of entries applied to each stand in this treatment as were applied in the planned thinning scenarios, meaning that the final density was much lower than the planned thinning scenarios.

4.2.8 FVS settings

The time period modeled was from 2008 to 2120. The 112 year period was selected so that metrics such as basal area and density by species would be calculated by FVS for each time

step from 2010 to 2110 as these metrics are calculated at the beginning of each time step by FVS. These metrics were not calculated for 2120. Only results from 2010 to 2110 were used in subsequent analyses. The standard time step was 10 years but, an additional cycle boundary was added in 2015 so thinning treatments could be applied in five year intervals between 2010 and 2020, since FVS only applies treatments and produces output at the beginning of each time step. FVS automatically updated the 2008 inventory data to the first time step boundary in 2010. In doing so, FVS also produced output for the 2008 and 2010 conditions.

The maximum stand density index (MaxSDI) was set by species. The same MaxSDI was used for all site series. The value of MaxSDI was determined by finding tree density in a plot with a quadratic mean diameter (QMD) of 25.4 cm (10 in) on the stand density management diagrams for natural stands of western hemlock, coastal Douglas-fir and western redcedar (Farnden 1996). The MaxSDI for bigleaf maple was taken to be equal to that of Douglas-fir because no stand density management diagram was found for this species. No other species was found in the Freda plots that were modeled.

Adjustments to the height growth settings were based on data collected from Stanley Park. Height growth was modified to conform to the observed site index by using the predicted heights of the four largest diameter Douglas-fir trees per plot in the overstory layer in 2008. The observed site index was calculated by taking the FVS-estimated height of the 4 largest trees in each plot and graphing this on Douglas-fir site index curves for BC (Mitchell and Polsson 1988). The assumed age of the trees was 46 years in 2008. This assumed the trees were 1 year old and planted in 1963. It was also assumed the planted trees took 5 years to reach breast height.

Therefore, the assumed breast height age of Douglas-fir in the Freda stands was 41 in 2008.

After the estimated top height was graphed on the site index curves at age 41, the height growth curve for each set of 4 trees was then traced to breast height age of 50 years. The projected height at age 50 years was taken to be the site index of the plot.

Four trees with the largest diameters per plot were used to find the site index because these trees represent the largest diameter 100 trees per ha since the plots were 1/25 ha in size. The ratio of the observed site index for each plot and the default FVS site index for each plot was calculated. The FVS default site index is determined by the plant association. The ratio was input to FVS as a height growth correction and used for the entire modelling period (Table 4-3).

Table 4-3. Observed site index at Freda plots modeled and the height growth modification input to FVS to adjust the default site index to match the observed site index.

Plot	Site series / FVS plant association number	Observed site index	FVS default site index	Height growth modification
Dry low density	05 / 43	30.9 m (101 ft)	33 m (108 ft)	0.938
Dry moderate density	05 / 43	30.0 m (98 ft)	33 m (108 ft)	0.911
Dry high density	05 / 43	32.4 m (106 ft)	33 m (108 ft)	0.984
Wet moderate density	07 / 30	29.9 m (98 ft)	35 m (116 ft)	0.845
Wet high density	07 / 30	33.0 m (108 ft)	35 m (116 ft)	0.933

4.2.9 Hazard assessment

Long-term windfirmness was assessed by comparing the HDR for a given plot top height with the HDR of the high windthrow hazard class identified by Mitchell (2000a) (Fig. 4-2). Mitchell (2000a) shows windthrow hazard classes for single species stands superimposed on stand density management diagrams. The diagram for coastal Douglas-fir was used to find the high-hazard threshold for stands with top heights ranging from 14 to 35.5 m. This is the entire range for which Mitchell (2000a) indicates that enough data were present to make hazard assessments.

Short-term windfirmness should be maintained as well. Short term windfirmness refers to the windfirmness of the stand between thinning and the completion of post-thinning HDR adjustment found by Mitchell (2000b). This period lasts about 3 to 6 years. Recommendations from the literature were used to assess the modeled scenarios since it is not possible to predict the probability of windthrow using FVS.

4.3 Results

4.3.1 Removals

Under the planned treatment, the plots require 2 or 3 entries over 5 to 10 years to reach the target densities. The removal density and basal area differed from plot to plot (Table 4-4). Thinning changed the 2110 attributes of all plots (Table 4-5).

Table 4-4. Number, basal area, and species removed in each thinning entry for each plot.

Plot name and treatment	T1 _{year}	T1 _N	T1 _B	T1 _{species}	T2 _{year}	T2 _N	T2 _B	T2 _{species}	T3 _{year}	T3 _N	T3 _B	T3 _{species}
	A				A				A			
Dry low density (Plot 225)												
No treatment	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Planned thinning	2010	91	5.5	Fd – 100%	2015	67	6.1	Fd – 100%	N/A	N/A	N/A	N/A
Heavy thinning	2010	146	7.6	Fd – 100%	86	11.2	N/A	Fd – 100%	N/A	N/A	N/A	N/A
Dry moderate density (Plot 333)												
No treatment	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Planned thinning	2010	94	4.1	Fd – 74% Hw – 26%	2015	62	4.6	Fd – 100%	N/A	N/A	N/A	N/A
Heavy thinning	2015	148	7.6	Fd – 83% Hw – 17%	2015	84	8.7	Fd – 100%	N/A	N/A	N/A	N/A
Dry high density (Plot 341)												
No treatment	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Planned thinning	2010	151	5.7	Fd – 100%	2015	106	7.6	Fd – 100%	2020	77	8.0	Fd – 100%
Heavy thinning	2010	242	11.2	Fd – 100%	2015	138	13.5	Fd – 100%	2020	82	12.4	Fd – 100%

Table 4-4. cont.

Plot name and treatment	T1_{year}	T1_N	T1_B A	T1_{species}	T2_{year}	T2_N	T2_B A	T2_{species}	T3_{year}	T3_N	T3_B A	T3_{species}
Wet moderate density (Plot 226)												
No treatment	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Planned treatment	2010	109	5.3	Fd – 78% Hw – 22%	2015	79	8.0	Fd – 100%	N/A	N/A	N/A	N/A
Heavy treatment	2010	175	11.2	Fd – 86% Hw -14%	2015	101	13.1	Fd – 77% Hw – 23%	N/A	N/A	N/A	N/A
Wet high density (Plot 330)												
No treatment	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Planned thinning	2010	124	3.4	Fd – 100%	2015	89	4.6	Fd – 100%	220	64	5.3	Fd – 100%
Heavy thinning	2010	195	6.9	Fd – 100%	2015	114	8.5	Fd – 100%	2020	67	7.8	Fd – 100%

Key to symbols:

T1 _{year} - Year of 1 st thinning entry	T2 _{year} - Year of 2 nd thinning entry	T3 _{year} - Year of 3 rd thinning entry
T1 _{BA} - Number of trees removed per ha in 1 st thinning entry (N per ha)	T2 _{BA} - Number of trees removed per ha in 2 nd thinning entry (N per ha)	T3 _{BA} - Number of trees removed per ha in 3 rd thinning entry (N per ha)
T1 _{species} - Basal area per ha of trees removed in 1 st thinning entry (m ² per ha)	T2 _{species} - Basal area per ha of trees removed in 2 nd thinning entry (m ² per ha)	T3 _{species} - Basal area per ha of trees removed in 3 rd thinning entry (m ² per ha)
T1 _N - Species composition of stems removed in 1 st thinning entry	T2 _N - Species composition of stems removed in 2 nd thinning entry	T3 _N - Species composition of stems removed in 3 rd thinning entry

Table 4-5. Final (2110) plot attributes.

Plot name and treatment	N	Ht_{top}	QMD_{os}	D_{max}	BA_{tot}	BA_{species}	SDI	HDR_{mean}	HDR_{range}
Dry low density									
No treatment	150	56.7	72.4	104.6	61.7	Fd – 100%	326	73	55-85
Planned treatment	109	57.4	82.9	106.2	57.8	Fd – 100%	289	69	54-85
Heavy treatment	84	57.6	88.9	108.0	52.1	Fd – 100%	231	65	53-84
Dry moderate density									
No treatment	198	53.8	71.4*	98.3	60.5	Fd – 96% Hw – 4%	336	72 *	63-90
Planned treatment	170	54.3	80.2*	102.4	56.8	Fd – 99% Hw – <1%	312	62 *	55-85
Heavy treatment	142	50.6	87.7*	108.5	52.5	Fd – 89% Hw – 11%	291	62 *	49-84
Dry heavy density									
No treatment	195	54.4	64.1	97.5	62.8	Fd – 98% Cw – 2%	343	74	60-90
Planned treatment	121	55.5	78.0	99.1	58.1	Fd – 95% Cw – 5%	297	69	59-84
Heavy treatment	83	54.1	86.2	112.8	48.6	Fd – 91% Cw – 9%	224	63	49-72

Table 4-5. cont.

Plot name and treatment	N	Ht_{top}	QMD_{os}	D_{max}	BA_{tot}	BA_{species}	SDI	HDR_{mean}	HDR_{range}
Wet moderate density									
No treatment	163	54.9	70.0	113.3	62.6	Fd – 95% Hw – 5%	331	71	50-89
Planned treatment	117	55.4	80.6	116.8	59.6	Fd – 92% Hw – 8%	289	68	48-85
Heavy treatment	93	55.5	86.8	120.9	54.7	Fd – 100%	270	64	47-84
Wet high density									
No treatment	229	42.6	42.4**	76.2**	109.4	Cw – 72% Fd – 20% Mb – 8%	537	87**	64-98**
Planned treatment	120	42.8	52.8**	74.2**	121.0	Cw – 81% Fd – 11% Mb – 8%	519	86**	66-89**
Heavy treatment	70	46.2	60.3**	70.9**	127.1	Cw – 88% Fd – 4% Mb – 8%	509	85**	68-87**
* Fd only **Fd and Mb only									
Key to attributes:									
N – Trees per ha – residual trees only (N per ha)					BA _{species} – Percent basal area by species				
Ht _{top} – Top height (m)					SDI – Stand density index				
QMD – Quadratic mean diameter (cm)					HDR _{mean} – Mean height to diameter ratio				
D _{max} – Maximum diameter (cm)					HDR _{range} – Range of height to diameter ratios				
BA _{tot} – Total basal area (m ² per ha)					SS – Site series in the Coastal Western Hemlock Dry Maritime subzone				

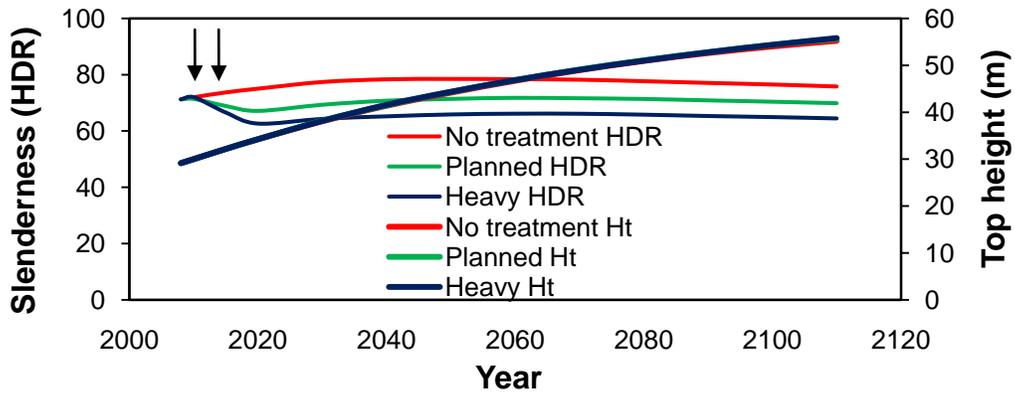
4.3.2 Slenderness

Thinning reduced slenderness in all plots (Table 4-5, Fig. 4-7 and 4-8). In all plots the heavy thinning treatment reduced slenderness more than the planned thinning treatment.

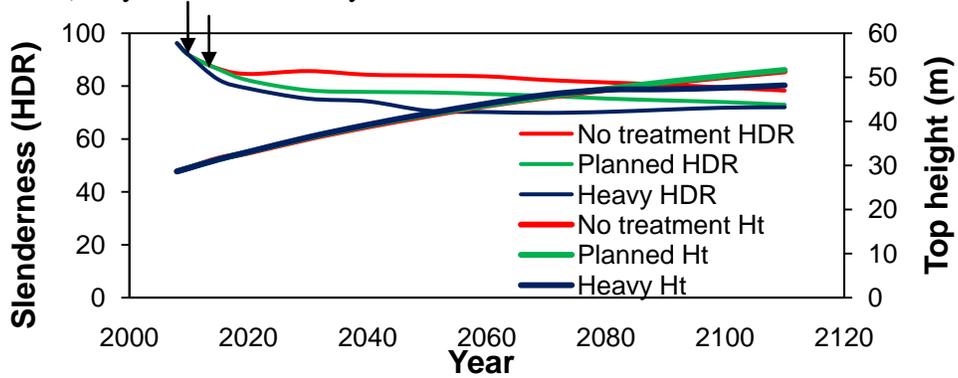
4.3.3 Windthrow hazard

Initially, 4 of the 5 plots were below the high hazard HDR threshold. The dry moderate density plot is initially above the high hazard threshold. Without thinning, the dry high density and wet high density plots would exceed the high hazard threshold by the year 2015 (Fig. 4-9). The other 2 plots may also grow into a high hazard state, but the heights of these plots exceeded the range for which hazard thresholds are defined (Fig. 4-9). Of the 3 plot that exceeded the high-hazard threshold without thinning, the planned thinning treatment was sufficient to keep stand slenderness below the threshold for the range in which the threshold is defined.

a) Dry low density



b) Dry moderate density



c) Dry high density

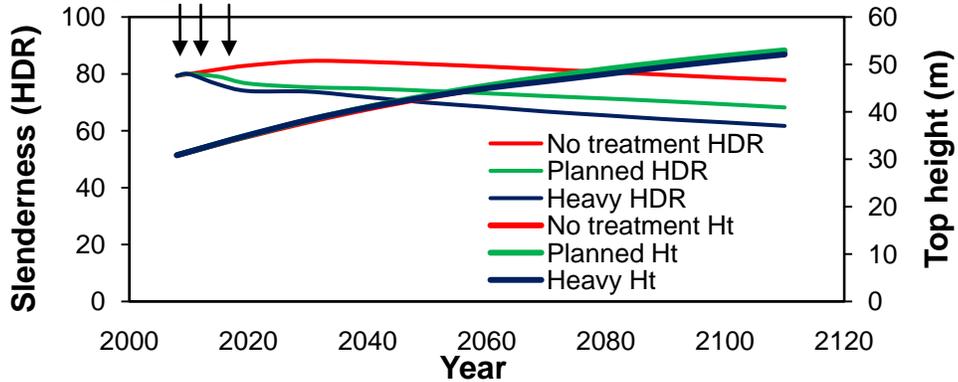


Figure 4-7 a-c. Slenderness and height over time of the overstory in each of the modeled plots in CWHdm site series 05. Arrows indicate thinning dates of the planned and heavy treatments.

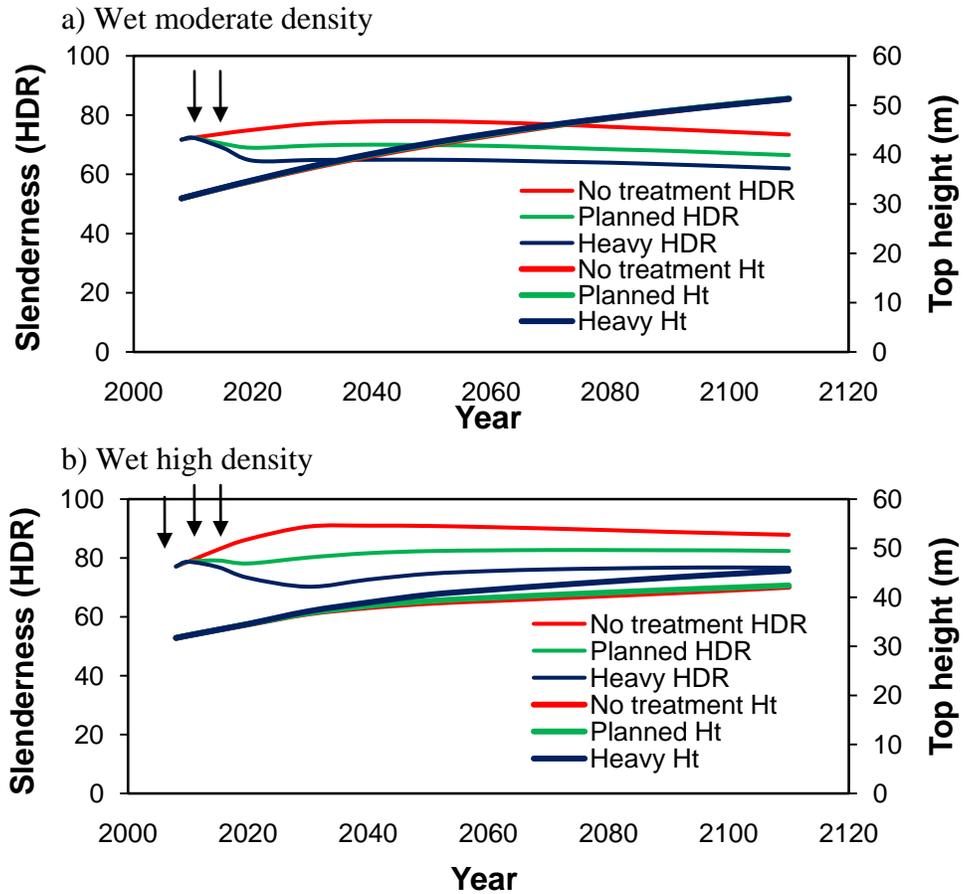
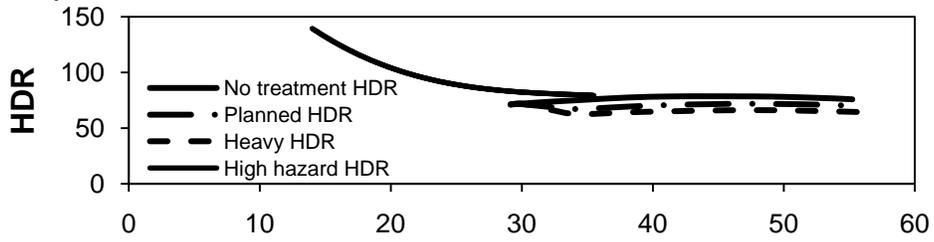
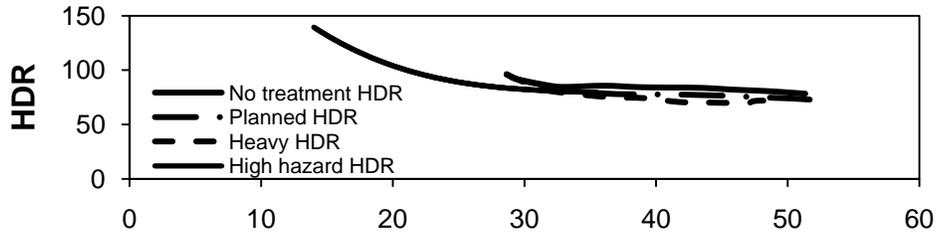


Figure 4-8 a-b. Slenderness and height over time of the overstory in each of the modeled plots in CWHdm site series 07. Arrows indicate thinning dates of the planned and heavy treatments. Note: beginning in 2030 fewer than 100 stems per ha were present in the wet high density heavily thinned plot.

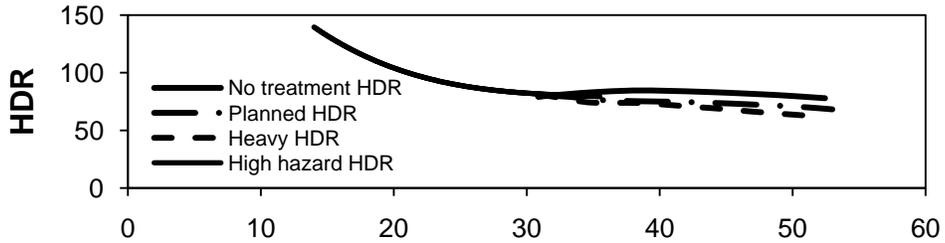
a) Dry low density



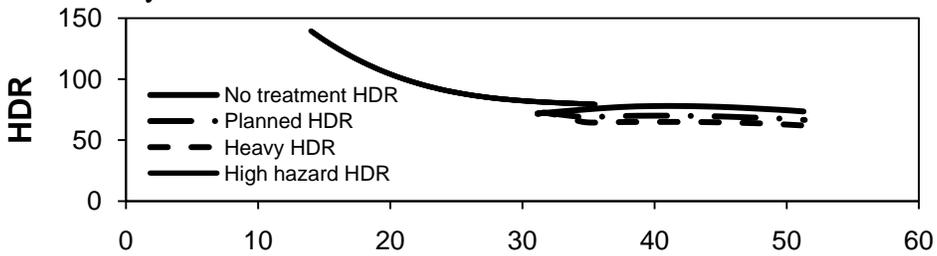
b) Dry moderate density



c) Dry high density



d) Wet moderate density



e) Wet high density

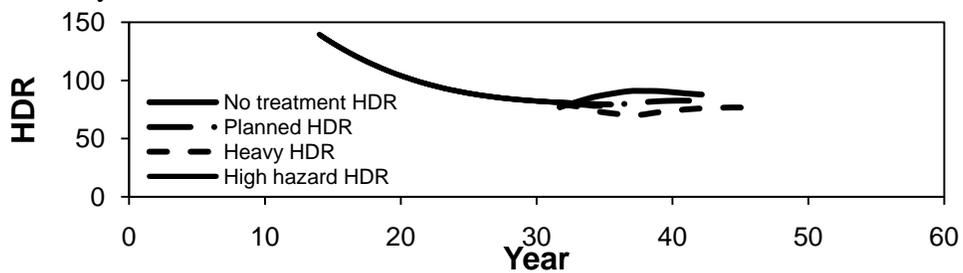


Figure 4-9. Top height and height to diameter ratio (HDR) of all modelled plots. Results from no treatment, and with the planned treatment and heavy treatment scenarios shown over a period of 102 years for a) dry low density plot b) dry moderate density plot c) dry high density plot, d) wet moderate density plot e) wet high density plot. Year is on the x-axis in all graphs.

4.3.4 Subcanopy response

The dry moderate density plot contained subcanopy trees at the time of inventory. These trees showed minimal growth in the no treatment and planned thinning scenario. In these scenarios, where canopy tree density by 2020 was 340 and 202 tree per ha in the no treatment scenario and the planned thinning scenario, respectively, the average understory tree height does not exceed 4 m over the modelling period. In the heavy thinning scenario, where the canopy tree density is reduced to 131 trees per ha and a basal area of 19.7 m² per ha by 2020 after two thinning entries, the understory trees respond and begin to grow toward the canopy (Fig 4-10).

The HDR of the subcanopy decreases over time in the no treatment and planned thinning scenario. In the heavy thinning scenario, the HDR first decreased but then increased after 2050 before stabilizing at about 90 by 2090. The HDR approached the high-hazard threshold in 2110 in this scenario (Fig. 4-11).

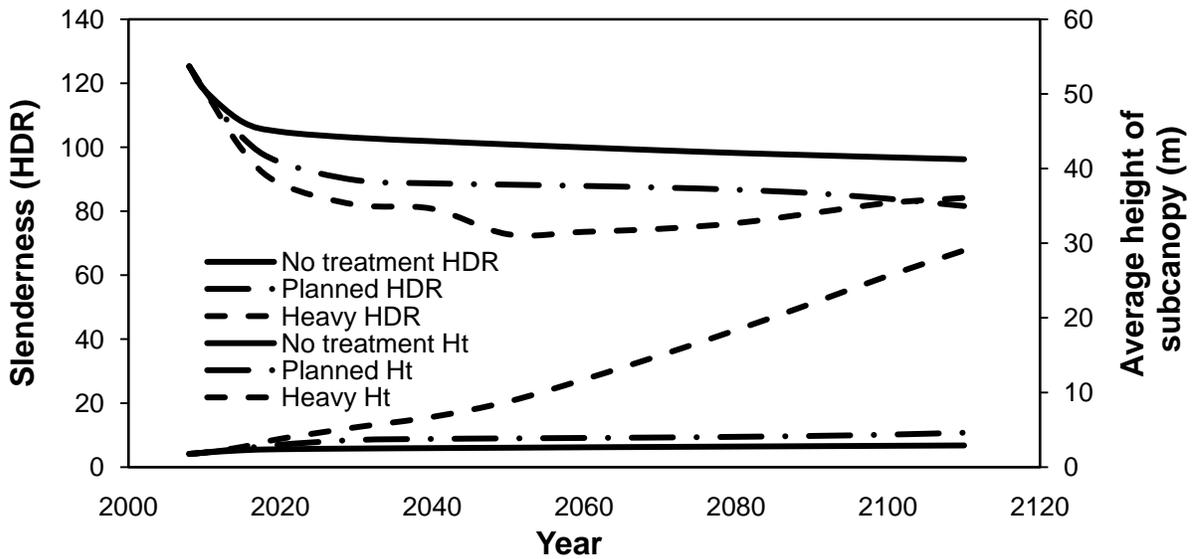


Figure 4-10. Slenderness and height over time of subcanopy trees in the dry moderate density plot for each of three scenarios. Note the scale on the left y-axis differs from similar figures above.

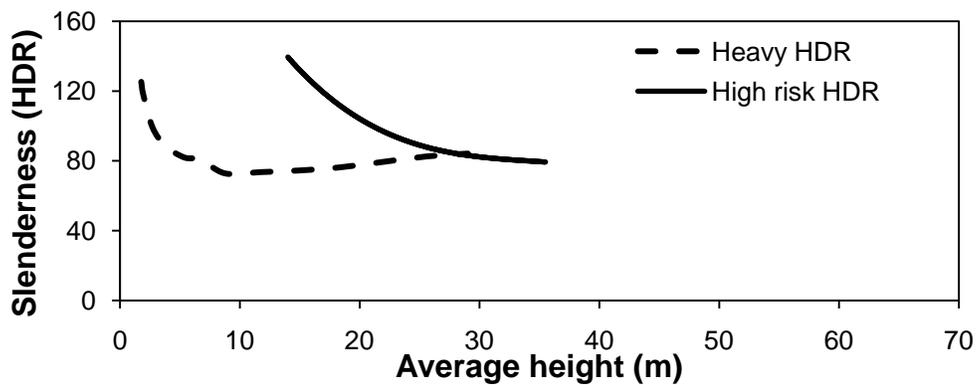


Figure 4-11. Average height (not top height) and slenderness of subcanopy trees in the dry moderate density plot in the heavy thinning option over a period of 102 years. Little growth is seen in the subcanopy trees in the other treatment scenarios and slenderness does exceed the high hazard threshold

4.4 Discussion

Slenderness decreases following thinning for two reasons. Thinning from below removes trees that are likely to be the most slender and so an immediate reduction in average stand

slenderness is seen. In the years immediately following thinning, residual trees change height and diameter growth allocation patterns (Mitchell 2000b). Mitchell (2000b) showed that this adjustment in growth allocation led to the reduction in HDR. Over time a new equilibrium pattern of allocation is reached. Finally, thinning opens growing space for trees, which can improve their growth rate leading to increased diameter growth, at least until crown closure (Oliver and Larson 1996)

The FVS model captures the effect of tree removal on average stand slenderness and the long term effects of increased growing space. It does not simulate the short term tree-level growth adjustments such as basal allocation and height growth reduction reported by Mitchell (2000b), which lead to the rapid reduction of slenderness following thinning.

In a single species stand with a single canopy layer, trees in lower canopy positions generally have smaller crowns because they continually lose their lower branches (Oliver and Larson 1996). Smaller crown size leads to less wood production (O'Hara 1988), and to changes in photosynthate allocation. Trees first allocate photosynthate to respiration (Waring and Schlesinger 1985). Excess photosynthate is then used for growth. The photosynthate used for growth is preferentially allocated to height growth over diameter growth, and this preference is enhanced in sub-dominant trees (e.g. Mitchell 2000b, Wonn and O'Hara 2001). As trees grow in size, an increasing amount of wood must be produced to maintain the diameter increment. Because trees cannot sustain ever increasing wood production at a rate to maintain diameter increment (Duff and Nolan 1957), even as height growth slows, stand-grown trees increase in

slenderness as they age. Healthy trees can adjust their growth allocation rapidly following thinning, and the most slender trees make the largest adjustments Mitchell (2000b) (Fig. 4-12)

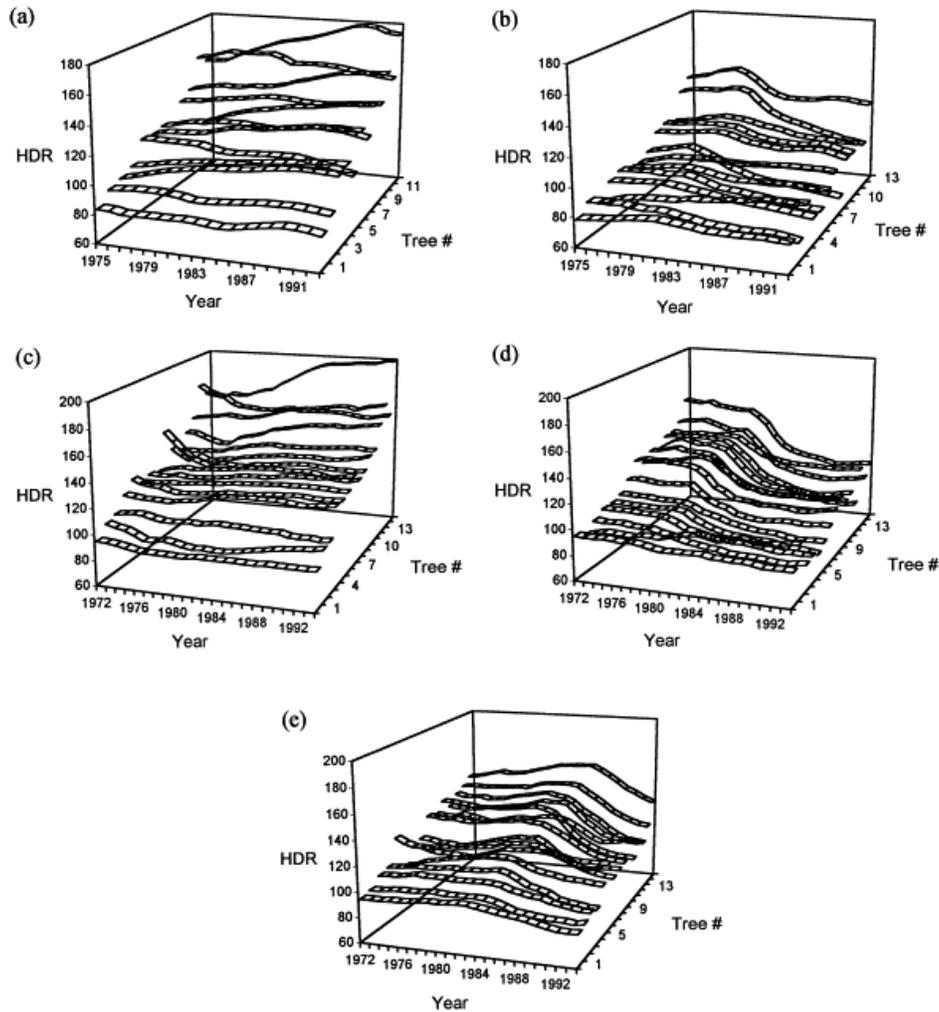


Figure 4-12. Annual height-diameter ratio (HDR) for each of the trees sampled by year. Trees are ranked along y-axis in order of HDR at the time of thinning with the most tapered trees in the foreground and most slender trees in the rear (a) Sitka spruce control; (b) Sitka spruce thinned in 1980; (c) Douglas-fir control; (d) Douglas-fir thinned in 1978; and (e) Douglas-fir thinned in 1893 (from Mitchell 2000b)

The effect of thinning on top height differed between plots. In 3 plots, there was little difference in top height between scenarios. In these 3 plots, the dry low, dry high, and wet

moderate density plots, the density after the final entry was between 121 and 151 trees per ha in the heavy treatment. Height growth of dominant trees is considered to be unaffected by stand density, and this is the reason that the dominant tree height is used to estimate site index (Oliver and Larson 1996). Since top height is calculated using the height of the tallest 100 trees per ha (40 per acre), the top height growth of a stand is unaffected by thinning provided the largest 100 trees initially remain the largest trees in the stand throughout the period of measurement. In 2 plots the 100 trees that had the largest diameters at the beginning were not retained in the thinning scenarios. Because of this, top height decreased in the dry moderate density plot and increased in the wet high density plot in the heavy thinning treatment scenario.

In the dry moderate density plot under the heavy treatment regime, top height increased more slowly in the last 30 years of the modeling period compared to the other dry moderate density scenarios. In this scenario, the density of the overstory after the final entry was 133 trees per ha. Continued mortality following thinning reduced this number further. When overstory density fell below 100 trees per ha, subcanopy trees were included in the calculation of top height which reduced the top height.

In the wet high density plot under the heavy thinning regime, top height growth in the exceeded that seen in the other wet high density plot scenarios. This was due to a species shift in the canopy from mixed Douglas-fir and bigleaf maple to primarily bigleaf maple.

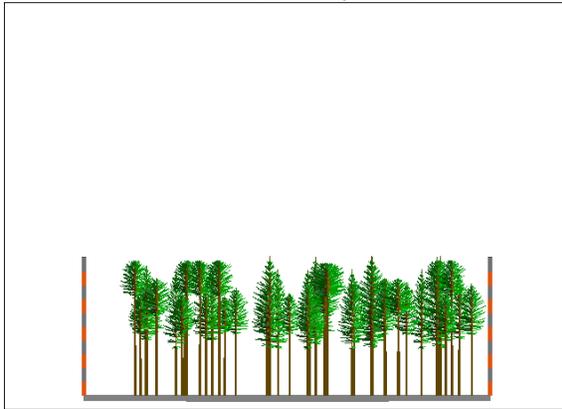
The dry moderate density plot currently exceeds the high hazard threshold, and the other four plots are approaching it. Examining the slenderness trends over time reveals the date at

which these plots will cross the high instability hazard threshold. The dry and wet high density plots will cross the high hazard threshold by 2020. The dry moderate density plot has an HDR higher than the high hazard threshold without thinning for the range in which the threshold is defined, and it is likely that the plot will remain in the high hazard zone based on the trajectory of the high hazard threshold beyond 35 m top height. Other plots appear to approach the high hazard threshold by 2030 and may do so but this happens at heights greater than 35 m. Given the current hazard status of Freda-era stands in Stanley Park, there is a window of opportunity between now and 2020 to treat stands in order to avoid having stands in the high windthrow hazard class for the next 20 to 50 or more years. However, since stands are already close to the high hazard threshold, thinning should be gradual.

A consequence of thinning from below is that the canopy height profile of the plots is raised and homogenized. Mason (2002) working with simulations of Sitka spruce plantations in the United Kingdom notes that irregular stands may be more windfirm on moderately exposed sites but not on the most exposed or protected sites. In relatively uniform stands, dominant trees are able to shelter shorter trees from the wind (Gardiner et al. 2000b). However, Gardiner (1995) suggests the strength of wind gusts may be reduced in stands that do not have a uniform canopy. Dominant expression in some Freda stands is poor, in others potential dominants are present (Fig. 4-13 a-b). In both cases, improving the growing conditions of the largest, most vigorous trees that have the greatest potential to become dominants would improve stand stability. This could be achieved by gradually removing subdominant neighbour trees.

a) weak dominant expression

Stand=225 Year=2008 Inventory conditions



b) strong dominant expression

Stand=333 Year=2008 Inventory conditions

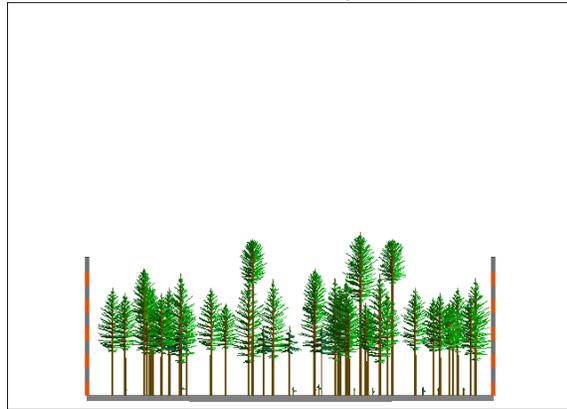


Figure 4-13 a-b. SVS profiles of two plots in 2008 showing a) weak dominant expression and b) strong dominant expression.

Thinning has been found to be an important factor in increasing short-term windthrow hazard. Recently thinned stands are more likely to experience wind damage than unthinned stands (Lohmander and Helles 1987). Scott and Mitchell (2005) similarly found that even non-contiguous canopy openings or thinned canopies within 300 m of a plot strongly increased windthrow occurrence. Gardiner et al. (1997) found that wind loading increased in proportion to inter-tree distance in thinned stands. If thinning immediate neighbours of dominant trees is under taken, the opening of small canopy gaps should not greatly affect stability. However, managers should not remove too many canopy trees with each entry. Depending on the number of dominants per ha, it might only be appropriate to remove one neighbour from every second dominant, for example, with each entry.

Targeted tree removals that open growing space around potential dominant trees may allow managers to retain higher stand densities than modeled here. This would be beneficial because trees in stands are able to mutually support each other. Gardiner (1995) found that

collisions between tree canopies dissipate wind energy. Milne (1991) found that contact between tree canopies reduced the displacement of sway. Following thinning, canopy collisions were reduced and tree sway increased (Rudnicki et al. 2003). By damping sway, tree-to-tree contact reduces the hazard of windthrow. To avoid short-term windthrow hazard by retaining mutual support structures Groome (1998) recommended not removing more than 30% of the trees in a stand.

No matter which of the 2 tested treatments is used, additional thinning entries may be required in the coming decades to further reduce stem density as remaining overstory trees continue to grow, or where there are subcanopy trees that are released and begin infilling the lower canopy.

The PN variant of FVS does not model regeneration aside from sprouting unless regeneration characteristics (species, density, and height) are specified by the user. Based on the observation that regeneration is establishing in 2006 storm damaged plots with retained basal areas of more than 40 m² ha (Chapter 3), it is likely that the both the planned and heavy treatment would create enough growing space to allow tree regeneration, and may release of existing subcanopy saplings. Western hemlock would be the primary species to regenerate due to its shade tolerance. Modelling results show, however, that subcanopy trees will not reach the canopy in the planned thinning scenario.

In the one plot where subcanopy trees were present at initial inventory, little growth was seen in the no-treatment and planned treatment scenarios. However, in the heavy treatment

scenario, the subcanopy grew from an average height of 2 m to 31 m by 2110, with the tallest trees reaching over 35 m. The HDR of these trees in 2110 was 90, placing them in the high hazard zone by 2090. If heavy thinning were undertaken, effort would be needed to keep the density of released and recruited trees low so that that stability is maintained. In addition, the recruitment of western hemlock is undesirable because the stand composition would move away from the target compositions for these stands.

4.5 Recommendations for management

Managers must balance long and short-term stability and costs of management when planning treatments for stands. They must also balance the desired stand structure with future windfirmness. In the most uniform stands the planned thinning regime may be too intense to maintain short-term stability. In the other stands, thinning from below will not provide enough growing space to achieve long-term stability targets. The targeted removal of trees around potential dominant trees may be a more effective thinning method to use. This method would minimize the loss of mutual support structures in the tree canopy while allowing future dominant trees to increase in windfirmness. The intensity of thinning should vary by stand, with the largest proportion of stems removed from the most differentiated, least slender stands.

Even with thinning, in all plots HDR stayed near the high hazard threshold. In the future, density management should occur earlier in the life of the stand to avert this problem. Wilson and Oliver (2000), for example, recommend that stands be thinned before they reach 10 m in height to achieve the best results for long term HDR reduction. They also note that once stands surpass 10 m in height, thinning will only slow the increase in slenderness of the 250 trees per ha

with the largest diameter. It should be noted this result from Wilson and Oliver (2000) differs from the trend of decreasing slenderness with increased height found on the SDMD in Figure 4-2. This is because Wilson and Olivier (2000) only consider the slenderness 250 trees per ha with the largest diameter, while the SDMD shows the slenderness trend for the entire stand.

The planned thinning to 200 to 250 stems per ha will hasten differentiation of the canopy, but it will not create new canopy layers based on the results from the single scenario with subcanopy trees. However, in stands with no subcanopy, tree regeneration will likely take place. Modelling results showed that subcanopy trees will grow into the canopy if overstory basal area is reduce to $< 20 \text{ m}^2$ per ha.

However, over the long term subcanopy trees will reach the canopy as low intensity disturbances removed trees from the canopy. In Stanley Park, this will lead to a gradual infilling of the canopy by western hemlock. To avoid this, managers should consider planting western redcedar in stands following thinning in gaps opened in the canopy. Planting should also occur following natural disturbances.

In the next chapter young stands will be modeled. These stands can be thinned before the 10 m threshold. The effect of this on stand structure and composition will be explored.

4.6 Limitations

Particular aspects of FVS limited its value when modeling the Freda stands. FVS does not model the effects of natural disturbances and diseases. It is likely that future disturbances

will change the forest in ways that are not accounted for by FVS. Individual trees or groups of trees may be killed by these future disturbances modifying stand densities and growth of remaining trees. FVS also does not model growth allocation changes made by trees following thinning as they adjust to new environmental conditions. The thinning scenarios presented here did not completely represent the prescription in Appendix 10 of the Stanley Park Forest Management Plan. It was not possible to model the spatial aspects of the prescription using FVS because it is a non-spatial modelling program.

Use of a model outside of the range for which it is calibrated can cause error. Growing conditions in the range for which the PN variant of FVS was calibrated may differ from those in Stanley Park. Calibration data came from National Forests from the central coast of Oregon to the Olympic Peninsula of Washington. The growth models for both small and large trees were generalized throughout the area which the PN variant covers but location effects are also taken in to account (Dixon 2002). Olympic National Forest is the nearest National Forest covered by the PN variant to Stanley Park. The northern edge of Olympic National Forest is located about 150 km south of Stanley Park. Stanley Park may differ from the stands used to calibrate the model in a number of ways included soil nutrients, precipitation and growing season length. One key climatic difference is that Stanley Park is situated near the mouth of the Fraser River and occasionally experiences strong cold winds from the interior as described by Stewart et al. (1995). Coastal forests in Washington and Oregon, with the exception of those near the Columbia River do not have river valley connection to the interior, so similar cold wind episodes are not experienced.

5. Post-2006 disturbance stand modelling

5.1 Introduction

The vision of the Stanley Park Forest Management Plan (Vancouver Park Board 2009) is to create and maintain a “resilient coastal forest with a diversity of native tree and other species and habitats”. To achieve this vision, park managers aim to create a range of forest stand conditions. Managing for resilience entails creating a forest that is able to withstand and recover from disturbance, pollution, and other stresses.

The windstorm in 2006 has provided an opportunity for park managers to significantly change the future structure and composition of the forest in ways that will allow target stand conditions to be met. They have modified the post-windthrow stand composition and probable natural trajectory by salvaging windthrown trees, preparing the site, planting and managing vegetation. Further modifications can be made in the coming years through density management. Management activities that happen soon after a disturbance can have great lasting effects on a stand. As Oliver and Larson (1996) note, early stand development has a profound influence on the long term composition and structure of the stand.

Managers of Stanley Park have noted several undesirable changes to structure and composition in park’s forest (Bill Stephen, pers. comm.). First, species composition is said to have changed over time. There is a perception that western hemlock has become a greater component of the canopy than at the time the park was founded. Bakewell (1980) notes this trend, and also reports an increase in deciduous canopy and a decrease in coniferous canopy. Neither of these changes was considered positive. Second, areas damaged in the 1962 storm and

subsequently planted with Douglas-fir have developed slender, single layer canopy stands with little species diversity (the subject of Chapter 4, above). Third, numbers of large diameter trees are thought to be in decline (Bakewell 1980). Finally, fire fuels have built up in some areas. Due to the proximity of the forest to structures and transportation routes in Stanley Park, as well as the proximity of park to downtown Vancouver, the consequences of a fire in the park are high.

Management activities can be used to resolve these issues in areas damaged in 2006 as well as undamaged areas. Modeling can inform managers as to what changes could occur under different thinning intensities in stands with different initial composition and site conditions. Modeling undertaken here focuses on the development of the 2006 windthrow stands.

Plots representing stands with a range of remnant overstory densities and site conditions were selected for modeling. In these plots, and throughout the area damaged in 2006, natural regeneration of conifers and deciduous species is taking place. Four conifer species were also planted in these areas to augment natural regeneration and to create more diverse forest stands. Future stand development in these plots was modelled with FVS with and without density management. Analysis of stand attributes at each time step reveals the development of key structural and compositional features including species diversity, tree size, slenderness and number of canopy layers.

Optimizing any one of these attributes in every stand in the park is not the vision of the Stanley Park Forest Management Plan. Instead a range of conditions should be created across Stanley Park to provide a diverse set of habitats and recreational spaces.

Modelling will be used to answer the following questions:

- 1) What will be the structure and composition of the 2006 windthrow stands in 100 years?
- 2) How will thinning treatments modify the structure and composition of these stands?
- 3) Under what management scenarios will target stand conditions be met in these stands?

5.2 Methods

5.2.1 FVS settings

Following the recommendation of the Forest Vegetation Simulator development staff, regeneration was entered into the simulation at 5 years of age (Erin Smith-Mateja, pers. comm.) This technique provided better results than entering regeneration at the time of disturbance because of inaccuracies within the small tree growth model of FVS. At this stage the planted trees in Stanley Park are expected to be above the height of most species of understory vegetation and 'free-to-grow'. Free-to-grow trees are tall enough that they are no longer in competition with surrounding non-tree vegetation for light.

Input data for the size and survival rates of natural and planted regeneration to age 5 are based on observations from regeneration studies at Malcolm Knapp Research Forest provided by Cheryl Power, R.P.F (Cheryl Power, R.P.F., pers. comm.). Natural and planted regeneration species densities are from field data collected in Stanley Park in 2009 as described in Chapter 3.

The year of disturbance was set at 2006. Regeneration was added to the scenario 5 years later in 2011. FVS modeled the growth of the regeneration from 2011 to the end of the simulation.

Regeneration data was input to FVS by entering the number of trees per acre for planted and natural regeneration > 30 cm to ≤ 130 cm tall by species. Percent survival is the percent of trees entered into FVS that survive to the end of the time step in which regeneration is entered. Survival percent and height at age 5 values, except for the percent survival of western hemlock, were from recommendations provided by Cheryl Power, R.P.F., from the Malcolm Knapp Research Forest. Percent survival for western hemlock was set slightly below that of other species based on observations by Ms. Power that mortality can be very high in areas where western hemlock regeneration is dense. This method assumes 100 % survival of regeneration from 2008 to 2011.

The same height and percent survival were used for both planted and natural regeneration for the 4 species of conifers that were planted (Table 5-1). This may underestimate the growth of planted trees as brushing crews removed competing vegetation from around these trees. The survival rate of planted trees may also differ from the value used but the direction of this difference is not known. The percent survival could be higher because brushing reduced competition, or lower because of accidental loss during brushing. Regeneration by sprouting in bigleaf maple results in larger trees of the same age (Cheryl Power, R.P.F., pers. comm.), however bigleaf maple regeneration from seed and from sprouting was not differentiated in user-defined regeneration. But, following thinning FVS automatically added sprouts for red alder, paper birch and bigleaf maple.

Table 5-1. Regeneration parameters for each species used in modelling the 2006 windthrow stands.

Species	Regeneration type *	Height at 5 years (m)		Percent survival regeneration to 2011
		CWHdm 05	CWHdm 07	
black cottonwood	Natural	2.4	2.6	100
bigleaf maple	Natural	2.4	2.6	100
bitter cherry	Natural	1.6	1.8	100
Douglas-fir	Planted/Natural	1.6	1.6	92
grand fir	Planted/Natural	1.2	1.2	92
paper birch	Natural	1.6	1.8	100
red alder	Natural	2.4	2.6	100
Sitka spruce	Planted/Natural	1.3	1.3	92
western hemlock	Natural	1.6	1.6	90
western redcedar	Planted/Natural	1.3	1.3	92

* 'Natural' indicates regeneration from seed, 'planted' indicates planted as part of the post-2006 restoration project

5.2.2 Calculation of stand level statistics

FVS was used to calculate stand level statistics including density, basal area, top height, stand density index, and height to diameter ratio. FVS was used to convert the US customary units for density and basal area into metric units. Other conversions were done in MS Excel. SDI and QMD were calculated as shown in Chapter 4, section 2.3, above.

5.2.3 Plot selection

Six plots were selected for modeling. The plots are evenly divided between Coastal Western Hemlock Dry Maritime Subzone site series 05 (herein, 'dry') and 07 (herein, 'wet'). The plots were all in located in areas mapped as having moderate or severe windthrow damage in

2006. Plots were selected to cover a range of residual overstory densities (Table 5-2; Fig. 5-1a-e). All trees greater than 1.30 m found in the plots in 2008 were considered residual trees (i.e. trees which survived the 2006 windstorm). SDI as used to determine the density of the residual overstory because it is the metric tracked by the mortality model of FVS. Density dependent mortality is applied by FVS when SDI reaches 55% of the maximum allowed SDI. The default maximum SDI, 950, was used in these scenarios.

While not a selection criterion, the plots also had a range of compositions of regeneration > 30 cm to \leq 130 cm tall (Table 5-3). Regeneration \leq 30 cm tall was not considered.

Table 5-2. Plot attributes in 2008. All values are for the residual trees only except for N_{tot} , the density residual trees plus regeneration > 30 and ≤ 130 cm tall.

Name	N_{res}	N_{tot}	Ht_{top}	QMD_{res}	BA_{res}	$BA_{species}$	SDI	HDR_{mean}	HDR_{range}	SS	Plot
Dry low residual	50	375	5.2	5.6	0.1	Cw – 17% Mb – 83%	2	92	72-112	5	352
Dry medium residual	326	4879	14.9	35.3	31.9	Hw – 10% Mb – 90%	224	76	21-88	5	107
Dry high residual	275	775	45.7	56.4	68.8	Fd – 91% Hw – 9%	401	73	46-102	5	38
Wet low residual	200	1776	4.0	3.6	0.2	Hw – 100%	3	106	81-146	7	303
Wet medium residual	126	951	33.8	64.3	40.4	Fd – 99% Hw – 1%	224	56	48-82	7	8
Wet high residual	525	875	44.8	38.6	61.9	Fd – 66% Hw – 34%	418	76	48-91	7	28

Key to symbols:

N_{res} – Number of residual trees per ha (N per ha)

N_{tot} – Number of residual trees including regeneration > 30 and ≤ 130 cm tall

Ht_{top} – Top height (height of 100 largest diameter trees per ha)

QMD – Quadratic mean diameter (residual trees only)

BA_{res} – Total basal area (residual trees only)

$BA_{species}$ – Percent basal area of each species (residual trees only)

SDI – Stand density index (residual trees only)

HDR_{mean} – Mean height to diameter ratio (residual trees only)

HDR_{range} – Range of height to diameter ratios(residual trees only)

SS – site series within the Coastal Western Hemlock Dry Maritime subzone

Plot – Sample plot number

a) dry low residual



b) dry medium residual



c) dry high residual



d) wet low residual



e) wet medium residual



f) wet high residual



Figure 5-1a-f. Photographs of 2006 disturbance plots modeled. The images are taken from plot center facing a cardinal direction. The units on the pole in the photographs are 10 cm.

Table 5-3. Attributes of planted and natural regeneration (> 30 to ≤ 130 cm tall).

Name	Planted trees (stems per ha)	Natural regeneration (stems per ha)	Total regeneration (stems per ha)
Dry low residual	Bg – 25 Ss – 75	Cb – 125 Dr – 25 Mb – 75	325
Dry medium residual	Cw – 25	Cw – 25 Ep – 50 Hw – 4428 Ss – 25	4553
Dry high residual	Cw – 125 Fd – 125 Ss – 50	Cb – 50 Fd – 25 Hw – 125	500
Wet low residual	Cw – 375	Cb – 50 Dr – 776 Ep – 100 Hw – 275	1576
Wet medium residual	Cw – 125 Fd – 125 Ss – 75	Cb – 125 Cw – 75 Ep – 125 Fd – 100 Hw – 50 Mb – 25	825
Wet high residual	Cw – 150	Cb – 50 Hw – 150	350

5.2.4 Layer identification

Canopy layers were identified in the 2110 profile view in the Stand Visualization System post-processor in FVS. Canopy layers were identified from the top down. The top-most layer ended when there was a gap between the bottom of the canopy of the shortest tree in the layer and the top of all shorter trees.

For the purposes of tracking layer slenderness over time, layers were identified as described in Chapter 4, section 2.6. Two layers of note are the advanced regeneration and regeneration layers. Subcanopy trees taller than breast height and under 9.1 m (30 ft) tall were identified as advance regeneration. Trees planted following the 2006 storm and natural seedlings found in 2009 were considered regeneration.

5.2.5 Management Scenarios

The base scenario is no treatment. Plots are projected without any treatments after the regeneration establishment stage. Thinning scenarios were modelled in cases where at least 1 canopy layer exceeded the high hazard HDR threshold or where stand structure or composition could be changed to better fit the target stand conditions found in the Stanley Park Forest Management Plan. Unlike the Freda stand modelling exercise, unique thinning regimes were created for each plot (Table 5-4). Unique thinning prescriptions were created because the modelled plots varied greatly in composition and structure. A single prescription would not have been appropriate for all plots.

Table 5-4. Thinning scenarios for each plot type.

Plot name and treatment	Year	Removal target	Thinning method	DBH Limit	Species retention/ removal preferences
Dry low residual					
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i> 1 st entry	2020	25% of stems	from below	no DBH limit	retention: Bg, Cw, Ss, Mb
<i>Heavy thinning</i> 1 st entry	2020	40% if stems	from below	no DBH limit	retention: Bg, Cw, Ss, Mb
Dry medium residual					
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i> 1 st entry	2015	residual density 990 trees per ha	from below	upper limit: 25 cm	retention: Cw, Ss removal: Hw
<i>Moderate thinning</i> 1 st entry	2015	residual 740 trees per ha, followed by removal of 60% of stems 5 to 20 cm DBH	from below, and throughout a diameter range	thin from below – upper limit: 25 cm; thin throughout a diameter range – lower limit: 5 cm upper limit: 20 cm	retention: Cw and Ss removal: Hw

Table 5-4. cont

Plot name and treatment	Year	Removal target	Thinning method	DBH Limit	Species retention/ removal preferences
<i>Heavy thinning</i> 1 st Entry	2015	residual 740 trees per ha, followed by removal of 85% of Hw stems 0 to 20 cm DBH	from below, and throughout a diameter range	thin from below – upper limit: 25 cm thin; throughout a diameter range – upper limit: 20 cm	retention: Cw and Ss removal: Hw
2 nd entry	2020	60% of Hw under 20 cm DBH	Thin throughout a diameter range	upper limit: 20	removal: Hw only
Dry high residual					
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i> 1 st entry	2015	40% stems	from below	upper limit: 25 cm DBH	retention: all species except Hw removal: Hw
Wet low residual					
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i> 1 st entry	2020	40% stems	from below	none	retention: Cw removal: Hw
<i>Moderate thinning</i> 1 st entry	2020	40% stems	from below	none	retention: Cw removal: Hw
2 nd entry	2030	40% stems	from below	none	retention: Cw removal: Hw

Table 5-4. cont					
Plot name and treatment	Year	Removal target	Thinning method	DBH Limit	Species retention/removal preferences
<i>Heavy thinning</i>					
1 st entry	2020	40% stems	from below	lower limit: 5 cm	retention: Cw removal: Hw
2 nd entry	2030	40% stems	from below	lower limit: 5 cm	retention: Cw removal: Hw
3 rd entry	2040	40% of stems of target species	through a diameter range	lower limit: 5 cm upper limit: 25 cm	only Dr and Ep removed
Wet medium residual					
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A
Wet high residual					
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i>					
1 st entry	2015	residual 300 stems per ha	from below	lower limit: 10 cm	none
<i>Heavy thinning</i>					
1 st entry	2015	residual 300 stems per ha	from below	lower limit: 10 cm	none
2 nd entry	2020	75% of Hw	through a diameter class	lower limit: 10 cm	none

5.2.6 Hazard assessment

Output from each trial was stratified by canopy layer then compared with the high hazard threshold for Douglas-fir stands as found in Mitchell (2000a). Identification of canopy layers is discussed above. The height and slenderness of all trees and post-disturbance regeneration found in inventory was tracked for the duration on the modelling period. Sprouts which developed after thinning were not tracked.

5.3 Results

5.3.1 Stems per ha

In 5 of the 6 plots, natural and planted regeneration accounted for most of the total density from establishment to 2110. Only the wet high residual plot had more residual trees than post-disturbance regeneration > 30 cm to \leq 130 cm tall.

5.3.2 Basal area

In the moderate and high residual plots, most of the basal area over the period of modeling came from the residual overstory. In the dry low residual plot, post-disturbance regeneration contributed more to total stand basal area than the residual trees from 2050 to the end to 2110. In the wet low residual plot, residual advanced regeneration sized western hemlock accounted for more basal area than the post-disturbance regeneration.

5.3.3 Removals

Thinning focused on post-disturbance regeneration and residual subcanopy trees. Few overstory trees were removed by thinning. Where overstory trees were removed, only the trees with the smallest diameters were taken. Because of this, the basal area removed by each entry was low. The most basal area removed by any entry was 4.4 m² per ha in the wet low residual plot (Table 5-5). In the dry low residual plot less than 1 m² per ha of basal area was removed. Density was reduced in most plots, in some cases greatly. In the dry medium residual plots over 3000 stems per ha were removed in the first thinning entries. The wet low residual plot was

unique in that thinning increased the number of stems per ha in 2110 as subcanopy trees survived in the thinned plot (Table 5-6).

Table 5-5. Number, basal area, and species removed in each thinning entry for each plot.

Plot name and treatment	T1_{year}	T1_N	T1_B A	T1_{species}	T2_{year}	T2_N	T2_B A	T2_{species}	T3_{year}	T3_N	T3_B A	T3_{species}
Dry low residual												
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i>	2020	91	<1	Cb – 100%	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Heavy thinning</i>	2020	143	<1	Cb – 100%	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Dry medium residual												
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i>	2015	3383	0.2	Hw – 100%	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Moderate thinning</i>	2015	3630	2.5	Hw – 100%	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Heavy thinning</i>	2015	4149	3.9	Hw – 100%	2020	47	0.2	Hw – 100%	N/A	N/A	N/A	N/A
Dry high residual												
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i>	2015	170	<1	Hw – 100%	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

Table 5-5 cont.

Plot name and treatment	T1_{year}	T1_N	T1_B	T1_{species}	T2_{year}	T2_N	T2_B	T2_{species}	T3_{year}	T3_N	T3_B	T3_{species}
			A				A				A	
Wet low residual												
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i>	2020	667	4.4	Hw – 54% Dr – 31% Ep – 8% Cb – 7%	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Moderate thinning</i>	2020	667	4.4	Hw – 54% Dr – 31% Ep – 8% Cb – 7%	2030	608	1.1	Dr – 82% Ep – 18%	N/A	N/A	N/A	N/A
<i>Heavy thinning</i>	2020	489	4.1	Hw – 59% Dr – 31% Ep – 10%	2030	188	2.1	Dr – 55% Hw – 40% Ep – 4%	2040	304	9.6	Dr – 93% Ep – 7%
Wet medium residual												
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

Table 5-5 cont.

Plot name and treatment	T1_{year}	T1_N	T1_B A	T1_{species}	T2_{year}	T2_N	T2_B A	T2_{species}	T3_{year}	T3_N	T3_B A	T3_{species}
Wet high residual												
<i>No treatment</i>	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Light thinning</i>	2015	109	1.4	Hw – 100%	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
<i>Heavy thinning</i>	2015	180	1.6	Hw – 100%	2020	77	3.0	Hw – 100%	N/A	N/A	N/A	N/A

Key to symbols:

T1_{year} – Year of 1st thinning entryT2_{BA} – Basal area per ha of trees removed in 2nd thinning entry (m² per ha)T1_N – Number of trees removed per ha in 1st thinning entry (N per ha)T2_{species} – Species composition of stems removed in 2nd thinning entryT1_{BA} – Basal area per ha of trees removed in 1st thinning entry (m² per ha)T3_{year} – Year of 3rd thinning entryT1_{species} – Species composition of stems removed in 1st thinning entryT3_N – Number of trees removed per ha in 3rd thinning entry (N per ha)T2_{year} – Year of 2nd thinning entryT3_{BA} – Basal area per ha of trees removed in 3rd thinning entry (m² per ha)T2_N – Number of trees removed per ha in 2nd thinning entry (N per ha)T3_{species} – Species composition of stems removed in 3rd thinning entry

Table 5-6. Plot attributes in 2110.

Plot name and treatment	N_{tot}	Ht_{top}	QMD	D_{max}	BA_{tot}	BA_{species}		SDI	HDR_{mean}[*]	HDR_{range}[*]
Dry low residual										
<i>No treatment</i>	254	44.0	56.1	102.4	62.7	Ss – 57%	Cw – 17%	367	103	45 – 171
						Mb – 14%	Bg – 6%			
						Dr – 6%				
<i>Light thinning</i>	193	44.2	64.3	102.4	62.7	Ss – 57%	Cw – 17%	347	83	45 – 170
						Mb – 14%	Bg – 6%			
						Dr – 6%				
<i>Heavy thinning</i>	183	44.7	66.5	10.24	63.6	Ss – 57%	Cw – 16%	347	70	45 – 147
						Mb – 14%	Bg – 6%			
						Dr – 7%				
Dry medium residual										
<i>No treatment</i>	2675	46.3	19.3	162.1	78.7	Hw – 60%	Mb – 40%	700	91	27 – 209
<i>Light thinning</i>	531	46.4	46.0	162.1	88.2	Hw – 68%	Mb – 31%	557	93	27 – 183
						Ep – 1%				
<i>Moderate thinning</i>	427	44.7	50.5	162.1	85.9	Hw – 61%	Mb – 33%	523	88	27 – 165
						Ep – 3%	Cw – 1%			
						Ss – 1%				
<i>Heavy thinning</i>	155	44.7	74.9	162.1	68.3	Mb – 55%	Hw – 55%	356	75	28 – 110
						Ep – 9%	Cw – 5%			
						Ss – 5%				

Table 5-6. cont.

Plot name and treatment	N_{tot}	Ht_{top}	QMD	D_{max}	BA_{tot}	BA_{species}		SDI	HDR_{mean}[*]	HDR_{range}[*]
Dry high residual										
<i>No treatment</i>	494	55.0	47.2	143.0	86.8	Fd – 85%	Hw – 15%	542	77	41 – 119
<i>Light thinning</i>	381	54.9	53.8	143.0	87.0	Fd – 85%	Hw – 15%	516	72	41 – 119
Wet low residual										
<i>No treatment</i>	582	48.5	46.2	94.2	98.2	Hw – 50%	Dr – 26%	619	79	54 – 177
						Cw – 17%	Ep – 7%			
<i>Light thinning</i>	687	46.3	41.7	96.5	93.6	Cw – 41%	Dr – 30%	615	76	54 – 96
						Hw – 20%	Ep – 8%			
<i>Moderate thinning</i>	1034	46.3	33.8	93.0	92.2	Cw – 43%	Dr – 30%	658	72	44 – 90
						Hw – 22%	Ep – 5%			
<i>Heavy thinning</i>	999	46.7	33.8	96.5	89.4	Cw – 48%	Dr – 19%	638	77	44 – 175
						Hw – 25%	Ep – 8%			
Wet medium residual										
<i>No treatment</i>	675	46.3	37.3	140.2	74.2	Fd – 94%	Hw – 6%	509	76	39 – 153

Table 5-6. cont.

Plot name and treatment	N_{tot}	Ht_{top}	QMD	D_{max}	BA_{tot}	BA_{species}		SDI	HDR_{mean}[*]	HDR_{range}[*]
Wet high residual										
<i>No treatment</i>	465	51.1	49.8	133.6	90.3	Hw – 53%	Fd – 47 %	553	81	40 – 124
<i>Light thinning</i>	426	51.5	51.3	131.6	88.4	Fd – 51%	Hw – 49%	534	72	40 – 99
<i>Heavy thinning</i>	361	51.9	54.6	134.4	84.7	Fd – 56%	Hw – 41%	500	69	40 – 99

* trees > 2.54 cm DBH only

Key to symbols:

N_{res} – Trees per ha – residual trees only (N per ha)N_{tot} – Trees per ha – residual and regeneration > 30 cm and > 1.30 m tall (N per ha)Ht_{top} – Top height (m)

QMD – Quadratic mean diameter (cm)

QMD_{res} – Quadratic mean diameter (cm)– residual trees onlyD_{max} – Maximum diameter (cm)BA_{tot} – Total basal area (m² per ha)BA_{speices} – Percent basal area by species

SDI – Stand density index

SDI_{res} – Stand density index – residual trees onlyHDR_{mean} – Mean height to diameter ratioHDR_{range} – Range of height to diameter ratios

SS – Site series in the Coastal Western Hemlock Dry Maritime subzone

5.3.4 Species composition and diversity

The two methods of thinning used in the trials, thinning from below and thinning through a diameter class, had different effects on stand composition. Thinning from below removed the smallest diameter trees. Where minimum diameter limits were not set, this thinning method removed the slowest growing individuals in the post-disturbance regeneration cohort. Typically, these individuals were all of the same species. For example, when light thinning treatment was applied to the dry low residual plot the effect was to reduce only the number of bitter cherry in the plot (Fig. 5-2 and Table 5-7). Heavy thinning completely removed the species from the plot. In both cases this had little effect on the basal area in 2110 (Table 5-6). The heavily thinned plot had only 0.9 m² per ha less basal area than the unthinned plot in 2110.

Lower DBH limits were put in thinning prescriptions in some plots to avoid removing – post-disturbance regeneration. With a DBH limit in place, FVS applied prescriptions only to trees within the size range eligible for cutting. In the wet high residual plot this was done to maintain regenerating western redcedar and Douglas-fir while removing only subcanopy western hemlock (Fig. 5-3).

Planting increased the number of species in 5 of the 6 plots. In addition, planting greatly increased the frequency of Sitka spruce and grand fir in the park. These species are currently uncommon in the park (Fig. 3-14), and no natural grand fir or Sitka spruce regeneration was found in any of the plots selected for modelling.

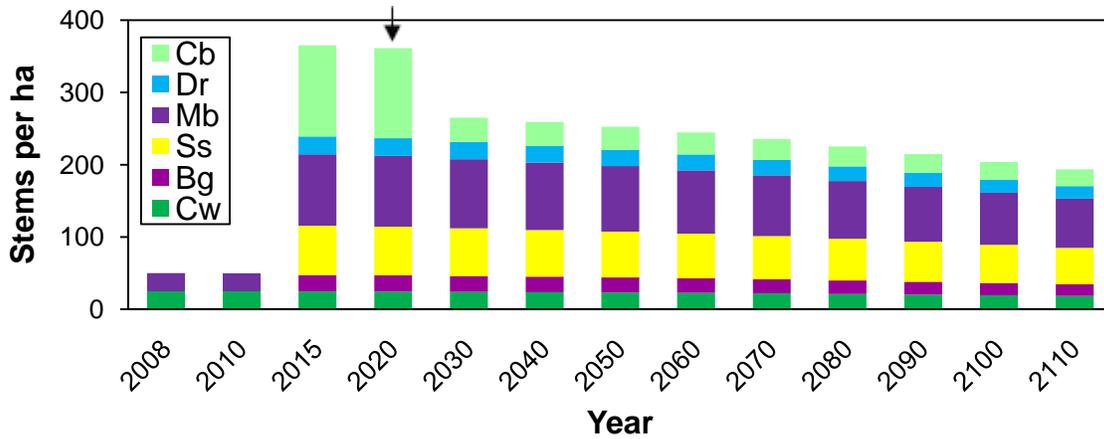


Figure 5-2. Stems per hectare by species at the end for each modeling time step from 2008 to 2110 for the dry low residual plot. Only bitter cherry was removed in this thinning scenario in which 25% of stems were removed from below. The arrow indicates the year of thinning.

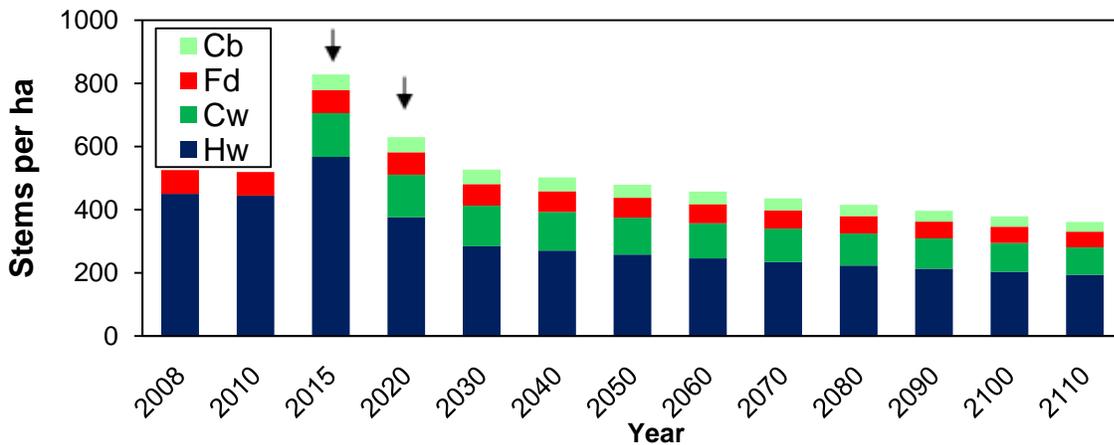


Figure 5-3. Stems per hectare by species at the end for each modeling time step from 2008 to 2110 for the wet high residual plot. Only western hemlock was removed in this thinning scenario in which the density of stems > 10 cm was reduced to 300 from below in 2015 and 75% of each size class of western hemlock > 10 cm were removed in 2020. The arrow indicates the years of thinning.

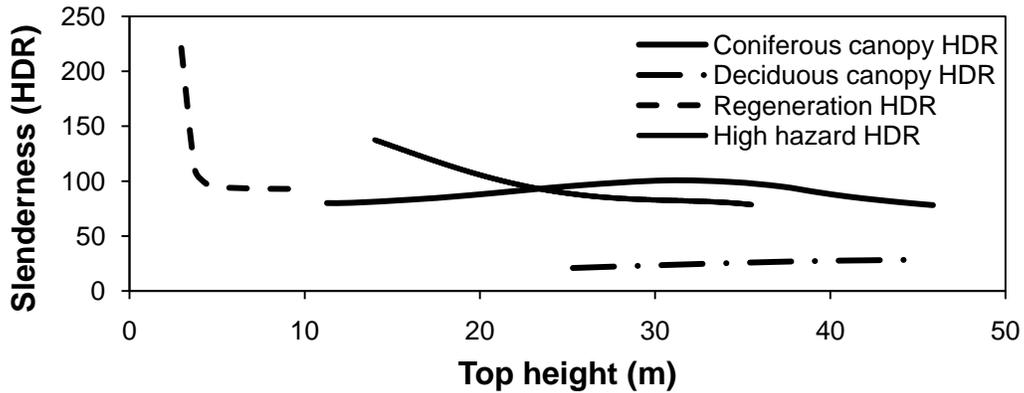
Table 5-7. Species found in each plot under each treatment regime. 'X' indicates presence of the species in 2110, '-' indicates species not present.

Plot name and treatment	Bg	Cb	Cw	Dr	Ep	Fd	Hw	Mb	Ss	Total
Dry low residual										
No treatment	X	X	X	X	-	-	-	X	X	6
Light thinning	X	X	X	X	-	-	-	X	X	6
Heavy thinning	X	-	X	X	-	-	-	X	X	5
Dry medium residual										
No treatment	-	-	X	-	X	-	X	X	X	5
Light thinning	-	-	X	-	X	-	X	X	X	5
Moderate thinning	-	-	X	-	X	-	X	X	X	5
Heavy thinning	-	-	X	-	X	-	X	X	X	5
Dry high residual										
No treatment	-	X	X	-	-	X	X	-	X	5
Light thinning	-	X	X	-	-	X	X	-	X	5
Wet low residual										
No treatment	-	X	X	X	X	-	X	-	-	5
Light thinning	-	-	X	X	X	-	X	-	-	4
Moderate thinning	-	-	X	X	X	-	X	-	-	4
Heavy thinning	-	X	X	X	X	-	X	-	-	5
Wet medium residual										
No treatment		X	X		X	X	X	X	X	7
Wet high residual										
No treatment	-	X	X	-	-	X	X	-	-	4
Light thinning	-	X	X	-	-	X	X	-	-	4
Heavy thinning	-	X	X	-	-	X	X	-	-	4

5.3.5 Slenderness

In plots where thinning was not sufficient to release subcanopy trees, thinning decreased or had no effect on the slenderness of each canopy layer. In some scenarios thinning released subcanopy layers which then increased in slenderness. For example, in the dry medium residual plot the regeneration layer reached only 10 m in top height by 2110 in the no treatment scenario (Fig 5-4). Under the light thinning treatment, which reduced the stand density to 990 trees per ha from below, trees in the regeneration layer were able to reach a top height of over 30 m by 2110 and while HDR increased from 79 in 2050 to 109 in 2090, when the top height of the layer was 28.5 m (Fig. 5-5 a-f and 5-6 a-f). Heavier thinning scenarios reduced the slenderness of this layer.

a) No treatment option



b) Light treatment option

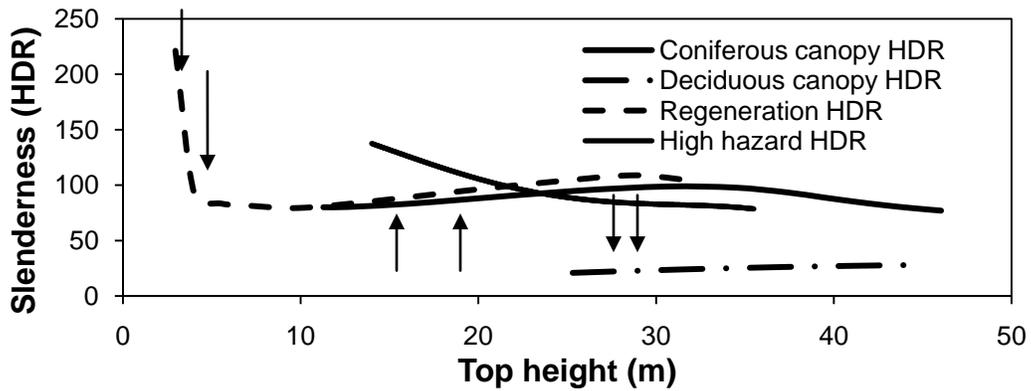


Figure 5-4. Slenderness (HDR) and top height (m) from 2008 to 2110 of advance regeneration and post disturbance regeneration in the dry medium residual plot a) without thinning and b) with light thinning. The high windthrow hazard threshold is represented by the high hazard HDR line. Arrows indicate the height when thinning occurred. The downward arrows points to the regeneration line and the deciduous canopy line, the upward arrow points to the coniferous canopy line.

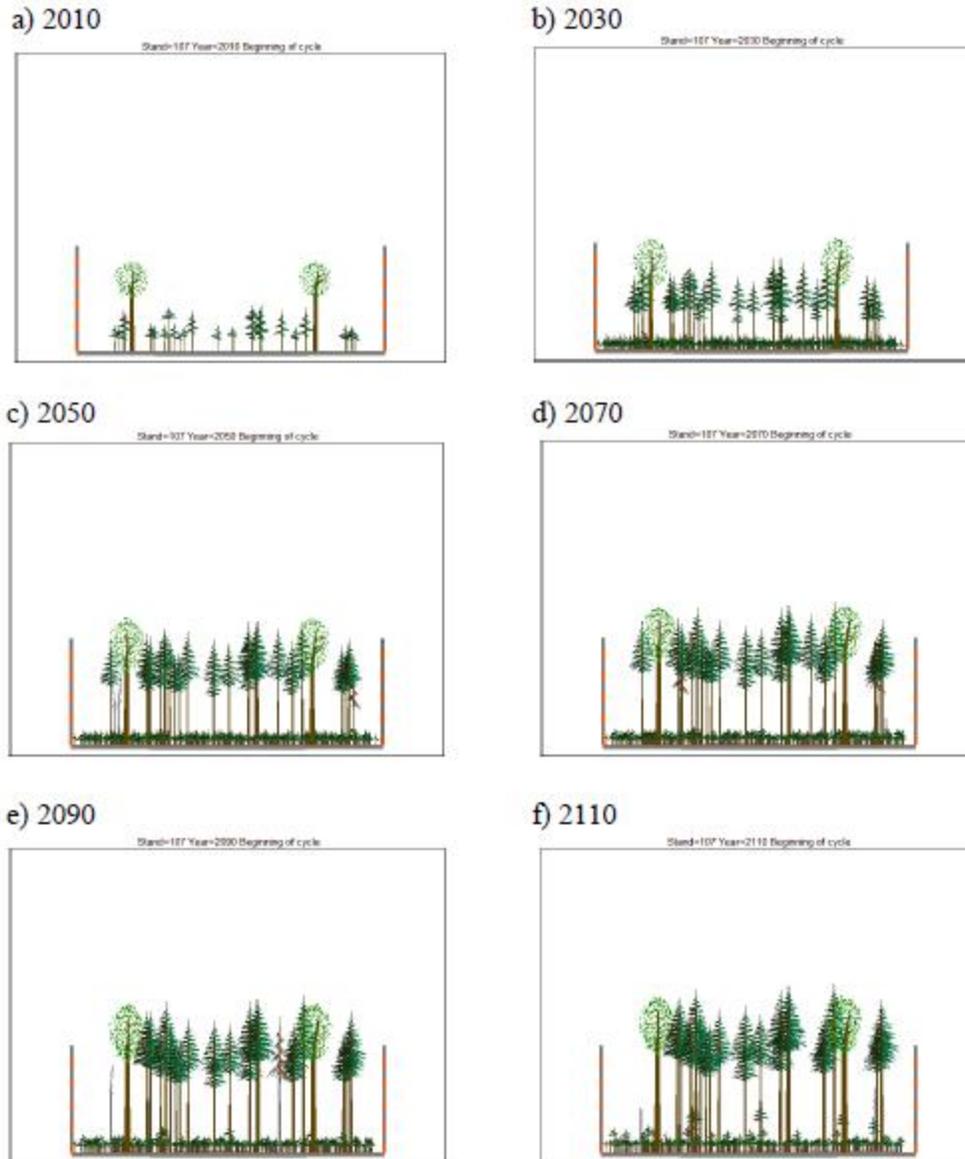


Figure 5-5 a-f. Profile views of the unthinned dry medium residual plot from 2010 to 2110. Red-crowned trees indicate trees that died in the previous 10 years. Note that little growth is seen in the regeneration layer which first appears in frame b. The vertical poles at the edges of the stands represent a height of 30.5 metres (100 feet).

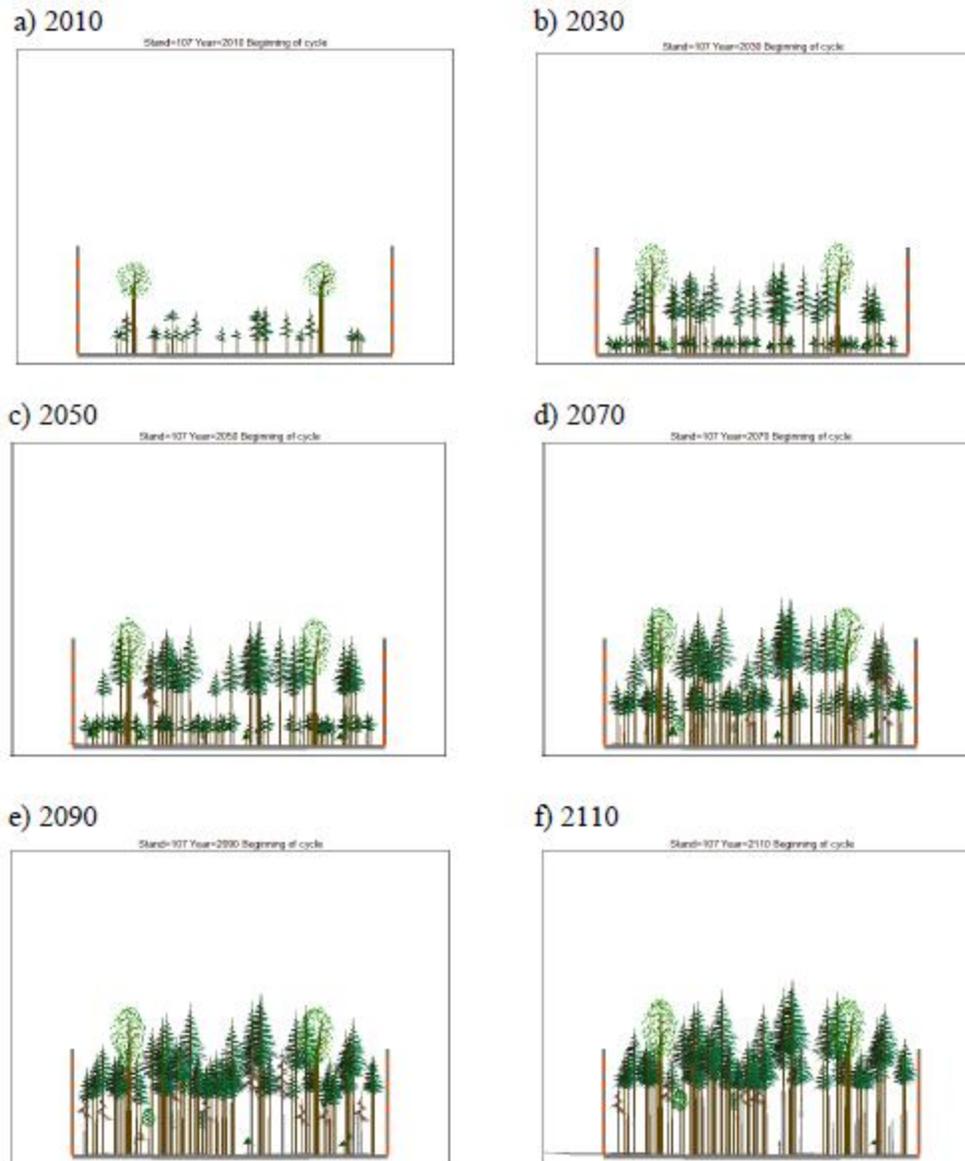


Figure 5-6 a-f. Profile views of the lightly thinned dry medium residual plot from 2010 to 2110. Red-crowned trees indicate trees that died in the previous 10 years. Note that the regeneration layer, which first appears in frame b, reaches the overstory by 2090 (frame e). The vertical poles at the edges of the stands represent a height of 30.5 metres (100 feet).

5.3.6 Tree size

Thinning increased the QMD in all plots except the wet low density plot. Thinning had little effect on the diameter of the largest trees in each plot. In the wet low density plot QMD

decreased with increased thinning intensity since the thinning regimes called for the preferential removal of western hemlock, regardless of size. Residual western hemlock trees with larger diameters than post-disturbance regeneration were therefore removed lowering the final QMD of the plot.

5.3.7 Number of canopy layers

Without thinning, each modeled plots developed 2 or 4 layers. The effect of thinning on the number of layers varied. In 3 of the 5 plots where thinning was applied, thinning treatments did not change the number of layers (Table 5-8). In the dry medium residual plot, the number of layers decreased from 3 to 1 as thinning intensity increased. Subcanopy layers were released and merged with the overstory (Fig. 5-5 a-f). In the wet low residual plot, the number of canopy layers increased from 2 to 4 with increased thinning intensity. Thinning released some but not all of the trees initially in subcanopy layers. These trees formed layers of intermediate heights.

Table 5-8. Number of canopy layers present in 2110 in each plot by for each scenario.

Plot name and treatment	No thinning	Light thinning	Moderate thinning	Heavy thinning
Dry low residual	2	2	-	2
Dry medium residual	3	2	1	1
Dry high residual	3	3	-	-
Wet low residual	2	3	3	4
Wet medium residual	2	-	-	-
Wet high residual	2	2	-	2

5.4 Discussion

The removal of the overstory by the windstorm in 2006 has given managers an opportunity to plant trees to change the species composition of the forest. However, in the disturbed areas western hemlock is the most common species in the seedling size class (> 30 to ≤ 130 cm tall). Residual sapling sized western hemlock trees are present as well. Management will be required to reach target stands conditions as the Stanley Park Forest Management Plan calls for mature stands of all types to have not more than 30% of the canopy occupied by western hemlock.

Regeneration data and modelling results show that western hemlock is the primary species to infill canopy gaps. This being the case, managers will need to constantly manage western hemlock densities to achieve the target conditions. Managers could reduce the need for such intensive management by allowing greater canopy cover of western hemlock. The 30% target was selected because western hemlock is considered less windfirm than other species. Some of this instability is due to the rooting locations of western hemlock on stumps and longs, not inherent structural weakness. Mitchell (2000a) shows that even-aged western hemlock stands have lower stand windthrow hazard than even-aged Douglas-fir stands of the same top height and quadratic mean diameter area. If western hemlock are found on secure rooting sites park managers should consider stands $> 30\%$ canopy cover of the species acceptable.

The simulations of the plots with low residual densities and the dry medium residual plot, which is composed mostly of pole sized trees, indicate that residual sapling (< 5 cm DBH) and pole-sized trees (≥ 5 to < 15 cm DBH), will dominate the future canopy. In the other medium

residual plot and the high residual plots, the remaining overstory dominates the plots throughout the modelling period.

Prior to disturbance, many pole and sapling sized trees were growing in high shade, diffuse lighting conditions well below the canopy, and so may have poor crown form. Oliver and Larson (1996) note that trees grown in high shade grow slowly and may become flat-topped. This is detrimental because this can lead to forking once these trees are released, which reduces stability of the stand (Mickovski et al. 2005). If residual pole and sampling sized trees have poor form they should be removed.

Removing poorly formed pole and sapling sized western hemlock from the stands by thinning within the next 10 years will allow regeneration to increase in relative basal area in two of the modeled plots. In the dry medium residual plot, heavy thinning allowed Sitka spruce and western redcedar to develop and reach 5% of the total basal area in the plot by 2110. Without thinning, these species represent less than 1% each of the total basal area in 2110. In this case, thinning is projected to create a stand that more closely resembles the target stand conditions than the unthinned stand.

Thinning in this plot allowed some subcanopy trees to grow into the canopy. This created a stand with more vertical heterogeneity but also one in which ladder fuels were present for a 40 year period (2050 to 2090). If subcanopy trees are allowed to grow into the canopy, managers must be prepared for a period of increased fire hazard before the trees reach the

canopy. One possible solution would be to vary thinning intensity through a stand so that fewer subcanopy trees reach the canopy.

In stands with high residual basal areas, such as the dry and wet high residual plots, regeneration will do little to change stand composition. Canopy openings will be quickly filled by existing canopy trees and tall subcanopy trees. Further reduction in overstory density by thinning the overstory would be required to open growing space for regeneration. Modelling showed that the overstory layers in these stands will not exceed the high windthrow hazard threshold so overstory thinning was not modeled. Light thinning of the subcanopy was modelled in the dry high residual plot, but no shift in the relative basal area of each species was seen.

The dry low residual plot differed from the other plots modelled in that no western hemlock or Douglas-fir was found in the sample plot. This plot had the lowest density of all the modelled plots. In 2110, Sitka spruce is projected to have the largest relative basal area in this plot. In the thinning scenarios modelled, the relative basal areas of the species in this plot changed little between simulations, but the slenderness of the stand was reduced.

In 4 of the 6 plots modeled, at least 1 canopy layer exceeded the slenderness threshold for high windthrow hazard at some point between 2008 and 2110. However, the residual overstory layer, the most exposed layer in all plots, was found to be below the high hazard threshold. Some residual overstory layers were taller than the range for which the high hazard threshold was defined. In these cases, the HDRs of the overstory would be below threshold values assuming the threshold follows same trajectory beyond 35 m top height.

Canopy layers that develop in partial or full shade are more slender than the uppermost canopy layers. Trees grown in shade have less photosynthate available to them than trees grown in full sun. Height growth is a greater priority than diameter growth in trees (Oliver and Larson 1996). Mitchell (2002) found that seedlings grown without mechanical stimulus grew more slender than those which were repeatedly bent. Seedlings grown in shade were more slender than those grown in full sun. The most slender seedlings were those grown in shade without mechanical stimulus. If this result also applies to sapling and pole size tree, overstory shading and the relative lack of bending stimulus from wind in the subcanopy will result in these trees developing high slenderness.

The slenderness of subcanopy layers should be of concern to park managers as trees from these layers may one day enter the overstory. The dry medium residual plot provides an example of a stand where residual trees previously growing in the subcanopy will achieve canopy level status. Under the light thinning scenario the regeneration layer reached a top height of 32 m and exceeded the high hazard threshold (Fig 54 b). Heavy thinning of both the residual trees and the regeneration was required to keep both layers from exceeding the high hazard threshold for instability.

In all the modeling scenarios, the residual large diameter trees had very low HDR ratios and were not a windthrow hazard. In Stanley Park, large diameter trees may not be in uniformly good condition. Some large residual trees are in poor health. A few large trees lost a significant number of branches in the windstorm and subsequent pruning efforts and may be at risk of death.

Dwarf mistletoe (*Arceuthobium tsugense* (Rosendahl) G.N. Jones) is common in the western hemlock in Stanley Park and is known to weaken western hemlock, making them more susceptible to windthrow (Muir and Hennon 2007). *Phellinus hartigii* ((Allesch. & Schnabl) Pat., a root rot, is also present and makes trees more susceptible to windthrow (Paul Lawson, R.P.F., pers. comm.). Large residual trees may also be in poor condition due to damage. Trees with wounds which subsequently decay are at increased hazard of bole breakage (Hennon 1995). Care should be taken to evaluate the health of large residual trees due to the damage or injuries they could cause if windthrown.

Thinning increased average tree diameter in all plots. Maximum diameter changed little between treatments. Increases in QMD were seen with increasing cutting intensity. These results were expected as thinning from below, the most commonly used treatment in the trials, removes the smallest diameter trees. In 1 plot QMD decreased with thinning intensity. In this plot the thinning regime preferentially removed western hemlock trees which were larger in diameter than other trees in the plot.

Larger average tree diameter is consistent with the vision of the Stanley Park Forest Management Plan. Thinning will hasten the development of stands with large QMDs. Thinning from below immediately increases QMD by removing the smallest diameter trees. Other thinning regimes may result in better diameter growth of particular individuals. For example, thinning around well expressed dominant trees could be a better thinning regime to use in order to increase the annual increment of these trees.

Large residual trees were maintained in the modelling scenarios. Large residual Douglas-fir and western redcedar trees remained the largest trees in the plots where they were found. Residual bigleaf maple was occasionally overtopped by conifers. In some cases, however, bigleaf maple grew to implausibly large size. In the dry medium residual plot, the model projected residual big leaf maple attaining 44 m in height by 2110. Uchyil (1989) notes that 15 to 21 m is the typical height range from mature bigleaf maple. Pojar and MacKinnon (2004) note that bigleaf maple can attain a height of 35 m.

The effect of thinning on the number of layers depended on the type and intensity of thinning. In the plots where the number of layers decreased with thinning, the reduction was due to intense thinning from below, which entirely removed the lowest subcanopy layers. The number of canopy layers increased in a plot where thinning from below and thinning through a diameter class were used to open gaps in a dense canopy thereby increasing growing space available to lower canopy layers. Thinning was particularly important to retain canopy layers in plots with open environmental conditions. In these plots the subcanopy layers merged with the overstory leading to plots with closed canopies and sparse subcanopy layers. Bauhus et al. (2009) suggest selection cutting of overstory trees to increase vertical canopy stratification. In the context of Stanley Park, this technique is appropriate where stability is not a concern, as canopy gaps increase short-term windthrow hazard (Gardiner et al 1997). However, creating a multilayered canopy increases the presence of ladder fuels and hence the fire risk.

Thinning in young stands to promote stratification will not be effective. Thinning small overstory trees in the dry medium residual plot ultimately led to a decrease in canopy layers

because subcanopy trees joined the overstory. Periodic overstory mortality will be required to maintain vertical heterogeneity in these forests. Small-scale low-intensity disturbance is the common mode of disturbance in coastal forests (Lertzman et al. 1996). If these disturbances are inadequate to create a forest with the desired level of heterogeneity, a thinning regime should be implemented.

An alternative to creating vertical stratification to increase stand and habitat heterogeneity in Stanley Park is to promote horizontal variation. Horizontal heterogeneity was a concern of forest managers at Stanley Park at the time of planting following the 2006 windstorm. Tree seedlings were planted in clumps for to increase horizontal heterogeneity. Franklin and Van Pelt (1994) note that horizontal variability is a key feature of old-growth stands. Bauhus et al. (2009) recommend group selection and variable density thinning to achieve this goal. These techniques could be used, particularly in the low residual stands, to promote horizontal structural variation. Without thinning, the low residual stands developed a two layer vertical structure with similar top heights.

The removal of overstory trees in 2006 allowed managers to change the composition of stands in Stanley Park. Planting successfully did this by adding species that were not found in natural regeneration in 5 of the 6 plots. Natural regeneration of deciduous trees added to the species diversity. All modeled plots had at least 4 species including 1 deciduous species.

The projected structure and composition of the post-2006 disturbance stands contrasts with the Freda stands in the previous chapter. Site preparation and planting of only Douglas-fir

in those stands led to the formation of low diversity stands. The Freda stands modelled, which were typical of stands with a high density of planted Douglas-fir, contained no more than 3 species. However, where disturbances or lower planting densities created more open canopies, species diversity may be higher. Retaining deciduous species as the 2006 windthrow stands develop should be considered by park managers to maintain tree species diversity in forest stands in Stanley Park.

5.5 Recommendations for management

- Select future dominants and thin early

The 2006 windthrow stands present an opportunity to forest managers at Stanley Park to make lasting changes to the forest composition and structure with little disruption to park users. In the thinning scenarios modeled, most trees removed were under 15 cm DBH, though the largest trees were 25 cm DBH. By thinning early, smaller trees will be removed and less equipment will be necessary. Removing small trees early in stand development is likely to be more acceptable to members of the public than later thinning because of this. It may also be less costly.

Trees with large crown ratios have more capacity to respond to openings in growing space following thinning than small crowned trees (Oliver and Larson 1996). In addition, smaller trees respond faster than larger trees following release (ibid.). Wilson and Oliver (2000) note that thinning Douglas-fir stands before they reach 10 m in height is most effective at reducing HDR. Thinning has been shown to increase windthrow hazard initially though retained trees are able to adjust to their new growing conditions reducing long term hazard (Achim et al.

2005). Thinning before trees reach 10 m in height also minimizes short term increases in windthrow susceptibility as stands are opened, because trees of this height are generally windfirm (Mitchell 2000a).

Freda stands, which are now over 45 years old with top heights between 25 and 35 m, provide an example of the management challenges posed by not thinning early in dense stands. Even the dominant trees in these stands have small crowns and so will respond slowly following thinning. Also, these stands are near the high hazard HDR threshold so short-term windthrow risk following thinning is great.

Maintaining well expressed dominant trees in each canopy layer should be the goal of forest managers. These trees have longer, more vigorous crowns and better growth form (straighter, few defects, no diseases) than trees in the same layer. In the simulations above, thinning from below was used to model this. In practice, maintaining dominants will require more than simply thinning from below. Current and future dominant trees should be selected as early as possible. Thinning should be used to open growing space around these trees.

- Where brush is a concern, use repeated light thinning

In areas where brush is a concern, light thinning entries carried out over a longer period could be used. In the modelling scenarios, regeneration was added when it was 5 years old and over 1.2 m in height. At this size the regeneration would be taller than most species of understory vegetation. Where salmonberry was a dominant understory species, trees might not yet be free to grow. Regeneration will be free to grow only when it no longer competes with the

understory vegetation for light. FVS does not factor in understory vegetation when simulating growth, so survival and growth rates of regeneration might be overestimated in the scenarios, particularly in the wet lower residual scenarios where salmonberry growth could be vigorous.

Thinning increases the cover of understory species (Bailey and Tappeiner 1998).

Excessive understory vegetation is undesirable to park managers. Excessive brush has long been a concern at Stanley Park, in the past because it did not fit with public perceptions of natural forests (Kheraj 2007), and currently because visitors have expressed safety concerns (Bill Stephen, pers. comm.).

Light thinning entries could be used to control understory vegetation while still achieving stand stability goals. The modeling scenario for the wet low density plot provides an example of this. In this plot, thinning was initiated in 2020 with 10 year return intervals. The target stand condition was achieved after three entries with removals of no more than 40% of stems of the target size and species. In this plot red alder and western hemlock were dominant. Thinning over a 20 year period in the heavy treatment scenario maintained a dense canopy which can reduce understory vegetation growth but still allow shade tolerant western redcedar to grow. In this simulation western redcedar had the highest relative basal area in this plot by 2110.

- Remove residual pole and saplings with poor form or of non-target species

Residual pole and sapling sized trees are projected to dominate the canopy of some stands. These trees are predominately western hemlock and some may have poor form due to past low light growing conditions, dwarf mistletoe or damage from the windstorm or subsequent

clean-up. Without removal, these trees will hinder the development post-2006 regeneration including the planted trees and western hemlock will continue to be a major component of many stands and will exceed target densities. Trees with poor form will also have reduced windfirmness.

This recommendation applies to western redcedar, a highly desirable species, only if a particular tree is deformed to extent that future growth will be greatly reduced or the tree could pose a hazard to visitors. Planting western redcedar or another desirable species in the gap left by the removal of a poorly formed western redcedar should be considered.

- Promote dominant trees in all canopy layers

Trees in subcanopy layers are sheltered from wind by taller trees. Even so, in Stanley Park, where partial disturbance by windthrow is a common type of natural disturbance, canopy and subcanopy layers should be managed to reduce the HDR. This can be done by identifying well expressed dominant trees in each layer and thinning to promote their growth. To do this park managers must remove trees whose canopy competes with the future dominant tree.

Thinning less vigorous trees in lower canopy layers is advantageous to thinning more vigorous trees in higher layers to create low slenderness, windfirm stands. Not all competitive trees must be removed at once. Thinning can be done over a number of entries. The scenarios above do not adequately simulate this type of thinning because the model used is non-spatial.

- Reduce western hemlock density

When thinning, managers should focus on removing western hemlock. In half of the 2006 windthrow plots modelled, relative western hemlock basal area exceeds target conditions in 2110 without thinning. An over abundance of western hemlock park-wide has been noted by others (Bakewell 1980; Kheraj 2007). Western hemlock is the most abundant species in the > 30 to ≤ 130 cm height class in the windthrow areas. In 1 of the modelled plots, the dry medium residual plot, the density of western hemlock trees in this height class was over 4400 per ha. In the same plot all other species accounted for only 125 stems per ha in the same height class. In some plots western hemlock regeneration will be over-topped by other species, but because of its shade tolerance it will remain in the subcanopy. Left in the subcanopy, western hemlock trees may create a fire hazard or be future windthrow hazard.

Thinning to maintain an open canopy and to allow subcanopy trees to develop may have undesired effect of increasing the fire hazard in the stand. This would occur if subcanopy structures such that ladder fuels are present. Ladder fuels carry a fire from the ground to the canopy. Additional thinning to remove subcanopy infill is one option for dealing with this possible fire hazard. The Stanley Park Forest Management Plan lists other methods for fire hazard reduction including chipping debris, removing slash, removing downed wood within 10 metres of flammable structures, pruning low branches and creating a 2 metre gap between the ground and ivy vines when removing them from trees.

In stands where enough growing space is present to allow for the development of a dense subcanopy layer, pruning is advantageous. Thinning the subcanopy may not be effective because following thinning a new subcanopy could develop in the opened growing space.

Pruning allows managers to retain a dense, but slowly growing sapling layer, while reducing connectivity between this layer and the canopy. Subcanopy thinning would only be effective where overstory canopy is increasing in density such that a new subcanopy would not develop following thinning.

- Identify areas of poor species composition and plan future management options

Managers can use thinning to change the composition and structure of stands. However, some stands will not reach target conditions. Three of the modelled plots are dominated by a pair of species in proportions outside the target stand conditions. These plots have moderate to high residual densities so planting is not a viable option because there is little open growing space. The overstory in these plots could be thinned to open growing space. However, this is an extreme action to take in stands where the problem is poor species composition. Future natural disturbances will provide an opportunity for managers to change the species composition of these stands through planting.

Underplanting below small canopy gaps has been used in the past by managers at Stanley Park. This management technique should continue to be used where possible in stands with poor species compositions.

- Monitor stands

To ensure that stands are not exceeding the high windthrow hazard threshold and that species composition is within target ranges, managers should regularly monitor stands within Stanley Park. Particular attention should be given to stands disturbed in 2006 over the next 10 to

20 years as the structure of these stands will change rapidly in that period. Monitoring surveys should also be done following thinning to ensure that the thinning activity achieved the managers' goals.

5.6 Limitations of data collection and modelling

In addition to the limitations of FVS noted in Chapter 4, other limitations were present in the methods used for data collection and assumptions of the Forest Vegetation Simulator introduce uncertainty into the results presented in this chapter. For example, any post-disturbance natural regeneration that was above 1.30 m in height by the 2009 was not counted, so some fast growing trees might have been missed. Additionally, stands and regeneration within Stanley Park were highly variable. Effort was made to capture the range of regeneration variability within each plot by sampling regeneration in 5 small subplots instead of 1 large plot. Even so, the full range of regeneration density or composition may not be captured by the survey data.

Plot level data was used to represent stands. In Stanley Park, low intensity natural disturbance and decades of small scale management interventions have created stands with high horizontal variability. This variability was not always be captured by the plots. Larger plots or more intense sampling would have better captured the variability, but these options were not feasible due to time constraints.

Results from the simulations done using FVS must be interpreted within the context of the assumptions of the program. The stands created in FVS are based on individual plots.

Heterogeneity in the stand in which a plot is located are not spatially simulated in FVS. Additionally, spatial heterogeneity within a sample plot is not retained by FVS. For example, FVS will not differentiate structural differences between a plot with an intimate mix of canopy layers and one where cohorts are clustered in groups across the plot. Edge effects are also not characterized by FVS. The dry low density plot was located adjacent to a stand with a much higher residual density. This adjacent stand is expected to have an effect on the development of the modelled plot (e.g. as a source of shade, seed and disease). These effects are ignored by the model, which assumes the adjacent stands are the same as the modelled plot. Likewise, the dry medium residual plot is located adjacent to a road (Stanley Park Drive), and so the subcanopy will develop in higher light levels than assumed by the FVS, until the edge closes in.

Applying spatial heterogeneity in the thinning treatments was not possible because FVS is a non-spatial model. FVS allowed only the following thinning options, thinning from below, thinning from above, and thinning through a diameter class. It was not possible to vary retention within a stand or to thin adjacent to particular dominant trees to promote their development. Also, individual trees growing in canopy gaps created within a modelled stand did not have a growth response that differed from more shaded trees of the same size.

Finally, mortality of small deciduous trees and bitter cherry of all sizes did not mirror actual conditions. In several simulations small deciduous shade intolerant species (i.e. red alder, bitter cherry, and paper birch) remained in subcanopy for decades. As these are shade intolerant species that are not typically found in the subcanopy (Esser 1995; Harrington 1990; Uchyil 1991) Bitter cherry also lived too long. Esser (1995) notes that the life span of bitter cherry is about 40

years. Bitter cherry between 70 and 100 years old are present in at UBC and at the Malcolm Knapp research forest, but in low numbers (Steve Mitchell, R.P.F. pers. comm.). In the modeling scenarios many bitter cherry remained present for over 100 years. In no plot was bitter cherry lost due to senescence.

5.7 Conclusion

The target stand conditions in the Stanley Park Forest Management Plan can be achieved by forest managers. Understanding the effects of early density and composition on future stand structure and composition is needed to ensure that proper thinning prescriptions are designed and implemented. The scenarios given in this chapter provide a view of one possible outcome from each of several thinning prescription alternatives. Actual stand development will differ from the modelled scenarios. The prescriptions used in the scenarios may be operationally unreasonable in some stands. Even so, they provide a guide to the relative intensity required to reach target stand goals. With sound management, the vision of the Stanley Park Forest Management Plan can be realized.

Over time, stands in Stanley Park will trend towards increasing canopy density and infill of canopy gaps by western hemlock. The trend will reduce the stand heterogeneity in the Park. This trend will only be reversed by management interventions or severe natural disturbance that disrupt the overstory, subcanopy and understory layers. As the consequences of a severe natural disturbance in Stanley Park are great, management interventions should be used whenever possible.

6. Conclusion

Urban trees live in habitats with a range of human interventions in forest processes and environmental conditions. The term 'urban forest' does not adequately take into account this range. Three terms, 'self-maintaining-forest', 'semi-self-maintaining-forest' and 'urban treescape', were proposed in Chapter 2 to better define different segments of the gradient from areas with fully functional forest processes to areas where human interventions greatly change forest processes.

Within urban areas, recognizing that a range of forest conditions situated along a gradient of the relative influence of human modification of ecological processes and environmental conditions are present, can improve understanding of the conditions in which urban trees live, and ultimately improve management of these forests. Of particular importance to management is an understanding of what interventions are needed to compensate for changes in ecological processes and what ecological processes are incompatible with urban infrastructure and land uses.

Stanley Park is one example of an urban forest in which forest processes have been modified by humans creating a semi-self-maintaining forest. All the processes have been modified to some extent but of modification of regeneration and disturbance processes is particular importance. The regeneration processes has been modified by the long history of planting desirable species in the park. The disturbance regime has been modified by the removal of hazardous and unsightly trees, the increase in fire ignitions, fire suppression following ignition, and density management.

Changing the Stanley Park forest from a self-maintained forest to a semi-self-maintained forest has had structural and compositional effects on the forest. Fire suppression, for example, has removed disturbances that would periodically open forest stands to widespread Douglas-fir regeneration. Removing this disturbance type has favoured regeneration of shade tolerant species which can regenerate following low intensity disturbance. Other human interventions in Stanley Park forest processes included, pruning, thinning and understory vegetation management. Ecological processes have also been modified by invasive species and pollution from sources such as vehicles and pet feces.

Because of both human intervention and natural stand development, some stands within Stanley Park differ in species composition and vertical structure from the target conditions of a 'resilient coastal forest' outlined in the Stanley Park Forest Management Plan. Two key issues are the uniform canopies found in stands planted following the windstorm Freda and excessive western hemlock density in some stands. Continuing management intervention will be required to reach the target conditions and to protect public and infrastructure from damage by falling trees or debris.

The by having vision of a 'resilient coast forest' managers at Stanley Park hope to create a forest that can withstand or recover quickly from disturbance. Natural disturbances will not be eliminated, and will remain an important driver of stand development. However, the damaged caused by disturbances can be reduced. Recognizing this, managers should make plans to guide response to future natural disturbances

As managers work to achieve the vision of the Stanley Park Forest Management Plan, forestry concepts should be applied to the management of the forest. For example, managers must take into account the impacts tree removal will have on the remaining trees in the stand.

As forest management activities are implemented natural processes will be modified. Managers will have to identify how natural processes will change and note that, while some natural processes may be incompatible with desired stand conditions, other natural processes that occur in Stanley Park are beneficial and should be maintained. Natural nutrient cycling and decay pathways can remove most of the debris from the forest floor, for example. Intervention in this process is only necessary in areas of particularly high volumes of debris or in areas at great risk of fire ignition.

Likewise, managers should note that management of stand conditions is not always beneficial. The Freda stands provide an example of this. Following the Freda windstorm, Douglas-fir trees were planted at excessive densities in the disturbed areas. These stands have developed into dense, single canopy layer stands with poor dominant expression. The stands are now approaching the high windthrow hazard level based on their slenderness and could cause great damage if windthrown.

As managers work to reach target stand conditions, they should remain aware of the history of forest structure and composition change in Stanley Park. Aerial photographs show that the severe wind storms opened growing space in 1934, 1962 and 2006 and changed the

composition of the forest canopy. In between these events, the canopy increased in density. Much of the park has remained conifer dominated, but mixed coniferous/deciduous stands have persisted in the northeast part of the park. Also, a stand south of Beaver Lake was changed from conifer to deciduous dominated following the 1962 storm.

The forest vegetation mapping results presented in Chapter 3 showed that 5 vegetation classes are present in Stanley Park, wet conifer, dry conifer, alder dominated, mixed coniferous/deciduous, and CWHxm. Up to 3 age classes are present in each vegetation class; young, mature and old mature. The Stanley Park Forest Management Plan defines target stand conditions for each of the possible 15 combinations of vegetation and age class. Natural forest development will change the structure and composition of these stands as they age.

Good management of these stands will require knowledge of the local ecology, defined goals and planning. Long-term forest planning, in particular, is necessary because of the long life span of forest trees and decadal scale of forest structural change. Modelling tools can be useful in planning management interventions and testing alternatives. If properly timed, management interventions can have very long lasting effects. For, example modelling shows that thinning entries in the 2006 windthrow stands in 2015 and 2020 will help to maintain stand HDR ratios below the high hazard threshold until at least 2110, to increase individual tree size, to maintain tree diversity and to promote multi-layered stand structures. These outcomes lead to stands which the better meet the vision of the Stanley Park Forest Management Plan than unmanaged stands.

Results from stand growth modelling can provide information to managers that may not have been apparent previously. For example, thinning was not effective at reducing stand slenderness in some situations. Where a subcanopy tree layer was present, thinning was shown to have the potential to release the subcanopy layer. This subcanopy layer could then grow to exceed the high hazard HDR threshold. Also, simulations of the 2006 blowdown areas showed that, where regeneration is occurring under a medium residual canopy, slenderness of post-2006 regeneration will exceed the high hazard HDR threshold. While actual stand conditions may differ from modelled results due to site conditions and disturbances, the results illuminate potential areas of concern. Managers can respond to these concerns by monitoring stand conditions, with particular attention paid to canopy layers projected to become overly slender. This in turn will allow them to better manage stands at Stanley Park to achieve target conditions.

Lessons from the Freda Stands should inform future management of the 2006 windthrow stands. Freda stand modelling shows that managers have waited too long to easily deal with stand instability problems. Early thinning is preferable for preventing excessive stand slenderness.

In addition, modelling shows that the composition of stands in some 2006 windthrow areas will not reach target specifications without management. Thinning within the next 10 years will be effective at changing stand structure and composition to reach target conditions. By thinning early managers will be able to retain conifers with high crown ratios, and remove excess trees while they are small. Small trees are more easily disposed of and can be removed with minimal impact on park users.

The operational benefits of early thinning can also be contrasted with the difficulties of thinning in Freda stands. Because of poor differentiation in these stands, dominants are poorly expressed. The trees slated for removal in thinning scenarios are up to 25 cm DBH, and over 20 m tall. Removing these trees without damaging surrounding trees and disposing of the trees in a way that minimizes fire and insect hazard poses operational challenges. Substantial impact on the forest floor is also likely if stems are removed from the stands. Public tolerance for thinning operations in the park is also low. Thinning the trees at this late stage will likely be less acceptable to the public than early thinning as the operations will have greater impact on trails and visually sensitive areas.

Data collection and monitoring of stands throughout the park should continue. Stand density and tree slenderness are key components to monitor, however other factors are of concern too. Dwarf mistletoe is common in the park and can weaken hemlock trees putting them at risk of wind damage. Incidences of root disease should also be tracked as affected trees are at risk of windthrow. Tracking the development of the forest and how closely composition and structure matches the target conditions can help managers decide on restoration strategies following future disturbances. Planting can be used to adjust stand composition over time as necessary where growing space is available.

Modeling software is an important tool managers can use to plan management, compare treatment and no-treatment scenarios and to communicate their plans and rationale to the public, administrators and politicians. The public has an interest in the management of the Stanley Park forest and other publicly managed forests but may lack knowledge of forest development and

disturbance processes. Stand visualization software can provide a graphic view of management alternatives proposed by forest managers. Combined with information about disturbance hazards such as windthrow and fire, these visualizations can be used to show how management alternatives can change forest structure and composition to achieve management goals.

Stand visualization is also useful for showing administrators and politicians the expected results of management and the value that these activities provide. In jurisdictions with constrained budgets, showing that management plans are effective at providing desirable outcomes is critical to securing funding.

Modelling results can inform the development and modification of prescriptions to reach long-term goals. Prescriptions can be tested using the model. If a prescription is found to be inadequate to reach long-term goals it can be changed prior to implementation in the field. By helping optimize the prescription, modelling can help managers create better cost estimates, minimize entries into the forest and better reach their goals.

Improved modelling tools would be of benefit to managers. The first 10 years following disturbance are of particular interest to managers at Stanley Park. In this period herbaceous vegetation and shrubs expand quickly as growing space is made available. Where deciduous trees are windthrown, sprouting is common. This vegetation creates a significant barrier to regeneration from seed of both conifers and deciduous species. This vegetation also is a competitor to planted trees. In the absence of good tools for modelling initial growth, it makes sense to continue monitoring changes by regularly remeasuring the permanent sample plots.

Assumptions had to be made about the initial development in the 2006 windthrow stands because of limitations of Forest Vegetation Simulator. These limitations were that FVS is not capable of projecting growth of understory vegetation and that the small tree growth model is poor. Survival of regeneration was based on estimates from managers at the Malcolm Knapp Research Forest who have experience in stands with similar soil moisture and nutrient regimes. It was impossible however, to take into account the effects of residual trees or the actual understory vegetation composition in these estimates. Manual brushing around planted trees may have also increased their vigour and height growth in their first few years which may allow planted trees to over-top natural regeneration, which under different management conditions would not necessarily be the case. A model which focuses on the interactions between tree regeneration and competing understory vegetation and which produces output in a format appropriate for use in FVS would improve the accuracy of this model for newly disturbed sites.

FVS could also be improved with additional data on the growth and mortality of shade intolerant deciduous trees. In the modelling scenarios presented in Chapter 5, shade intolerant deciduous trees remained in subcanopy layer. In the same scenarios, bitter cherry survived in the canopy for the a period of 102 years. This is longer than the typical lifespan of the species. Shade intolerant deciduous trees are relatively uncommon in coastal western hemlock forests such as Stanley Park compared to conifers (Green and Klinka 1994). Nonetheless, accurately modelling these species is important when considering species diversity and structural and compositional development in a stand.

Monitoring of the forest should occur after treatment. The data can be used to show that stands are developing toward the target conditions, or that additional treatment is necessary. Data can also be used to help managers prepare restoration guidelines following future disturbances. Finally, monitoring data can be presented to the public and policy makers to inform them of the state of the forest and show the progress being made toward achieving management goals.

The managers at Stanley Park have gone far in their effort to create a ‘resilient coastal forest’. The new Stanley Park Forest Management plan has long-term targets and outlines treatment strategies. The post-2006 management activities have started the disturbed stands toward these desired future conditions. Additional and sustained management interventions will be needed to fully realize the goal of a resilient, diverse coastal forest that is safe for heavy public use.

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Appendix 1 – SVS profiles of Freda stands

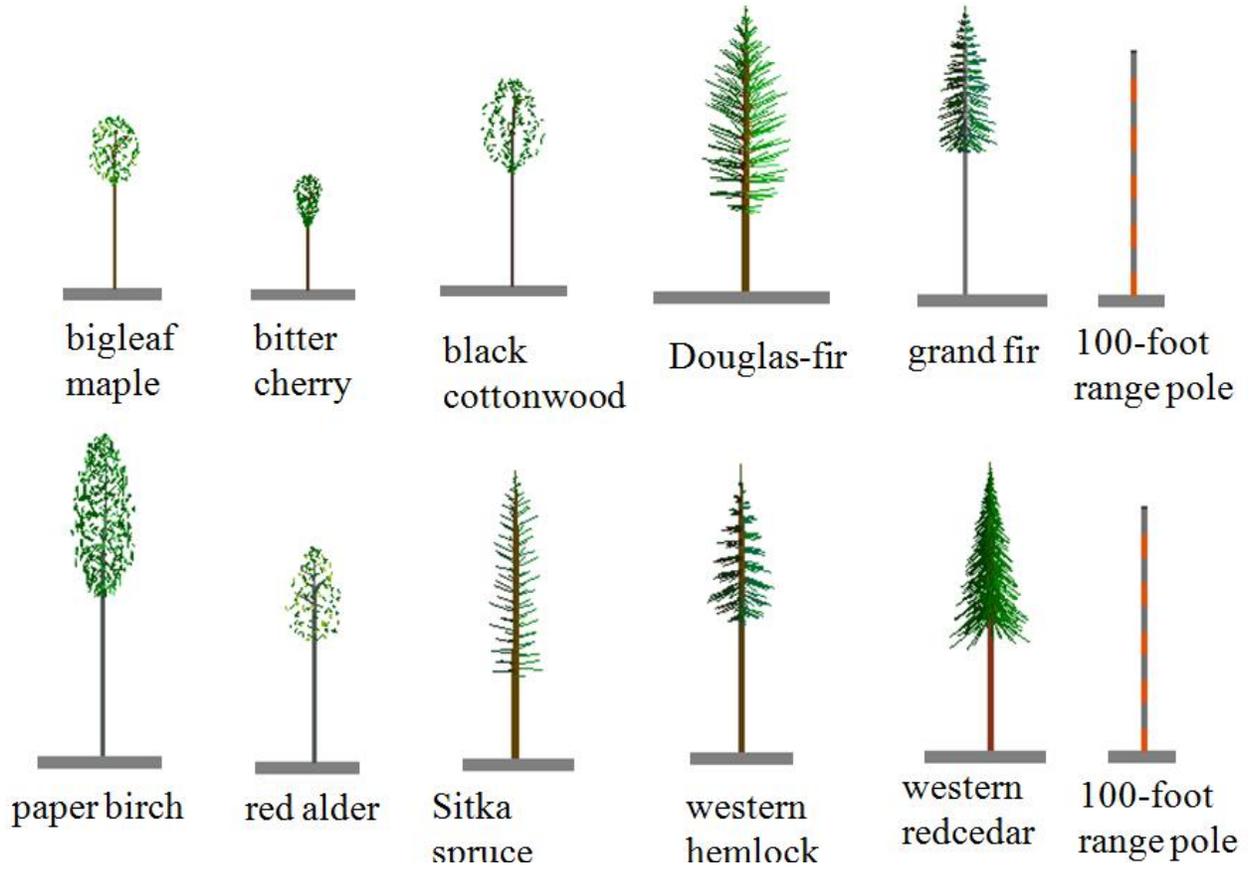


Figure A1-1. SVS profiles of the species present in plots modelled for analysis in chapters 4 and 5. The trees are presented here on the same scale at 70 years of age. The range pole is 100 feet (30.5 m).

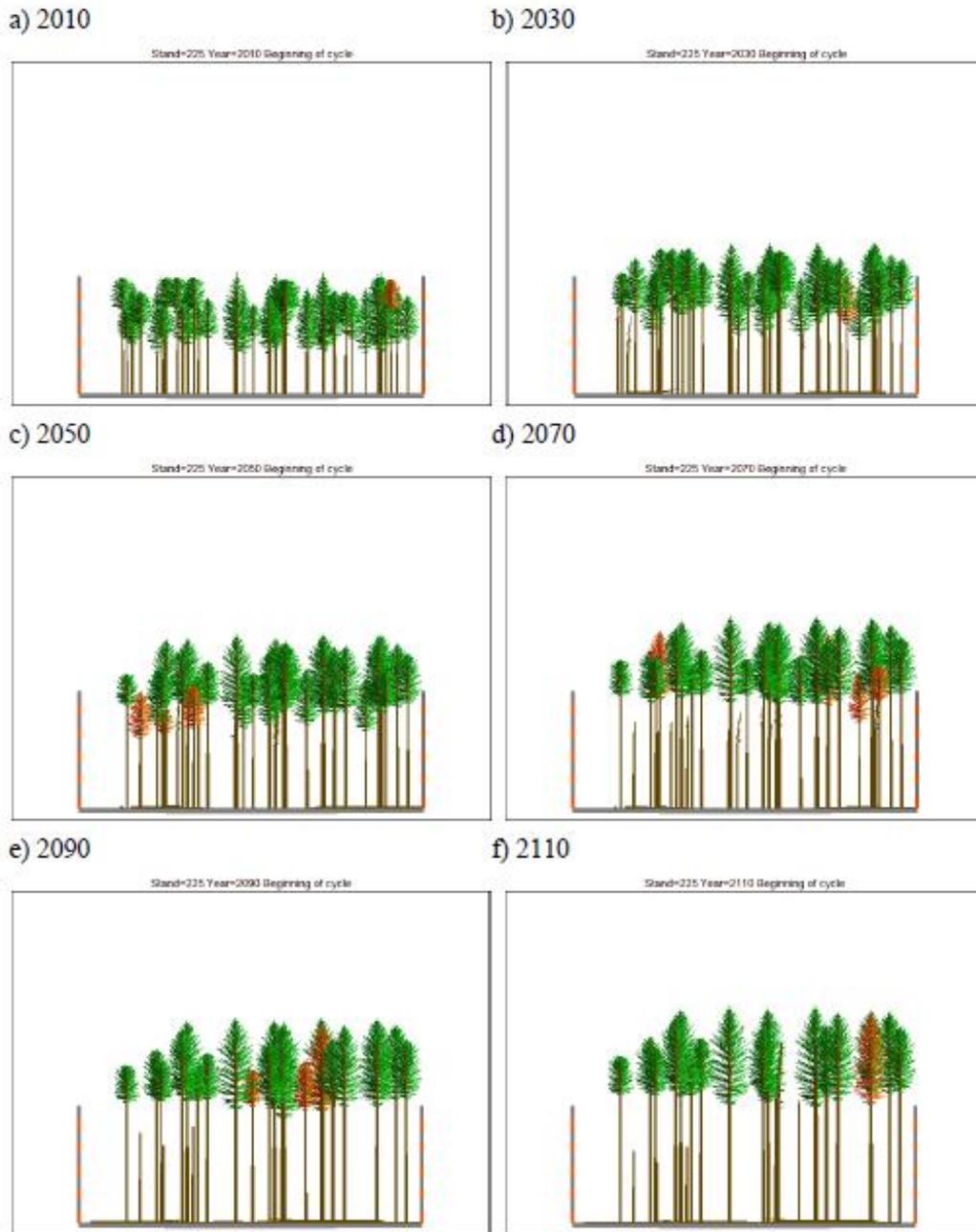


Figure A1-2. SVS profiles of the no treatment scenario in the dry low density Freda stand.

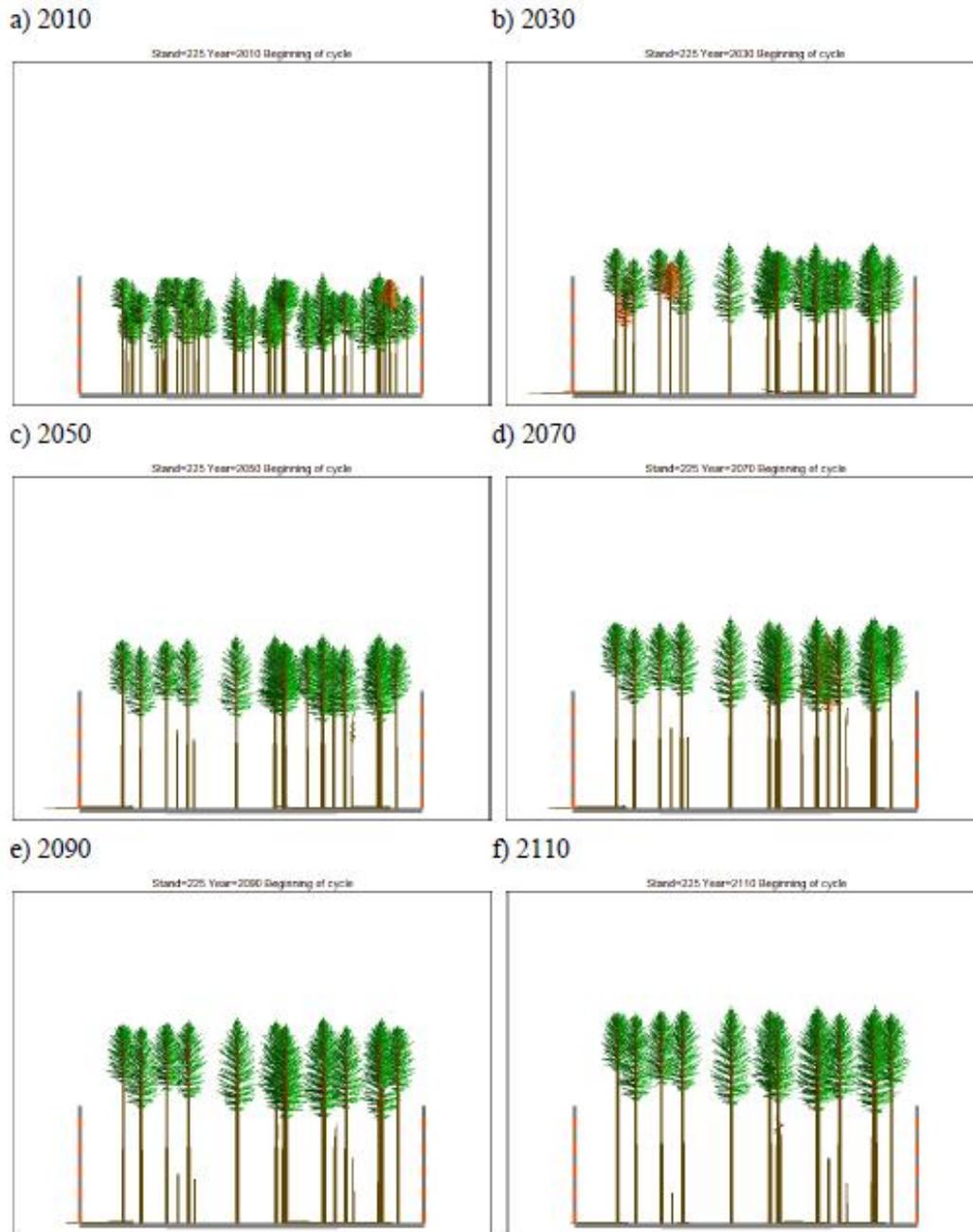


Figure A1-3. SVS profiles of the planning thinning scenario in the dry low density Freda stand.

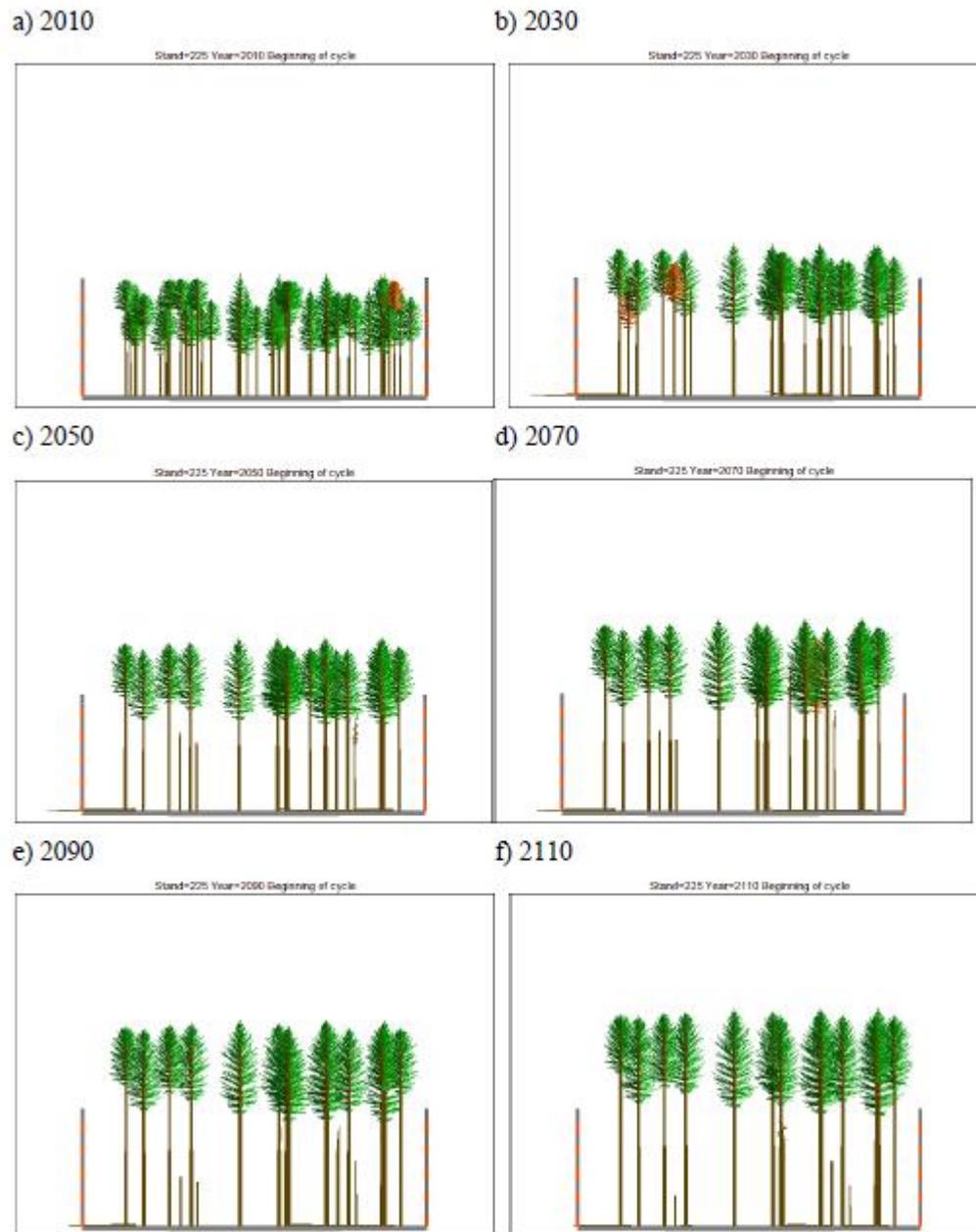


Figure A1-4. SVS profiles of the heavy thinning scenario in the dry low density Freda stand.

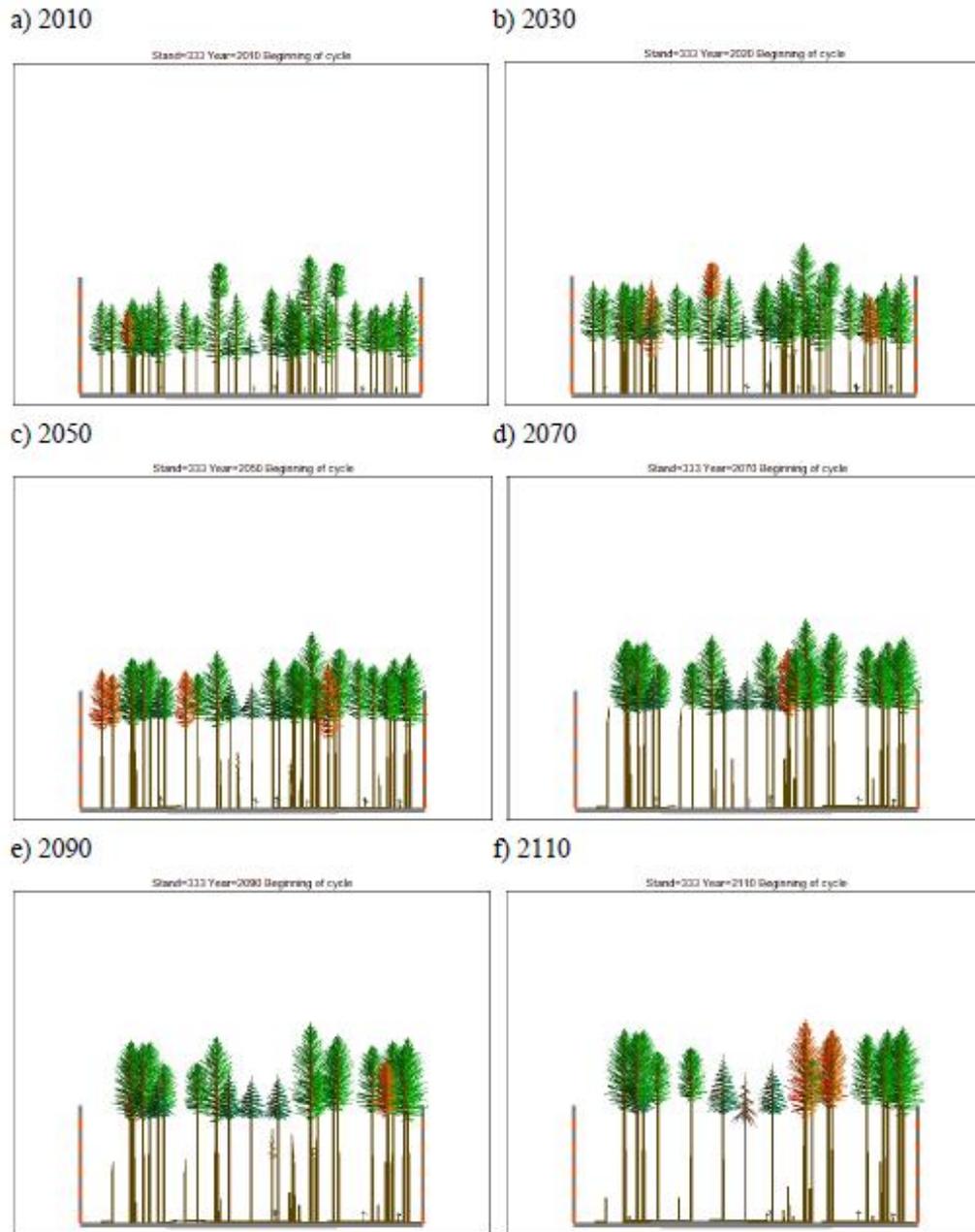


Figure A1-5. SVS profiles of the no treatment scenario in the dry moderate density Freda stand.

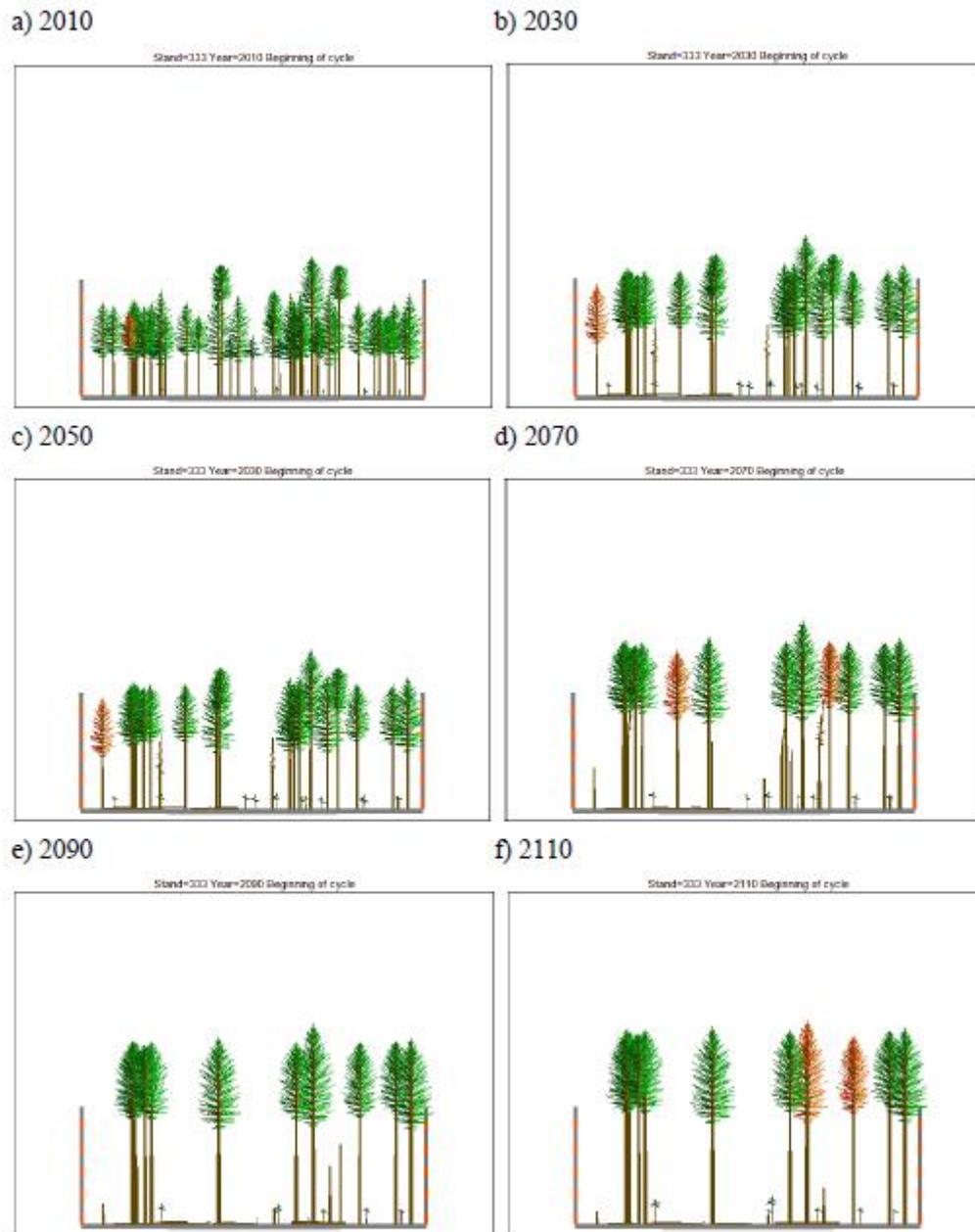


Figure A1-6. SVS profiles of the planned thinning scenario in the dry moderate density Freda stand.

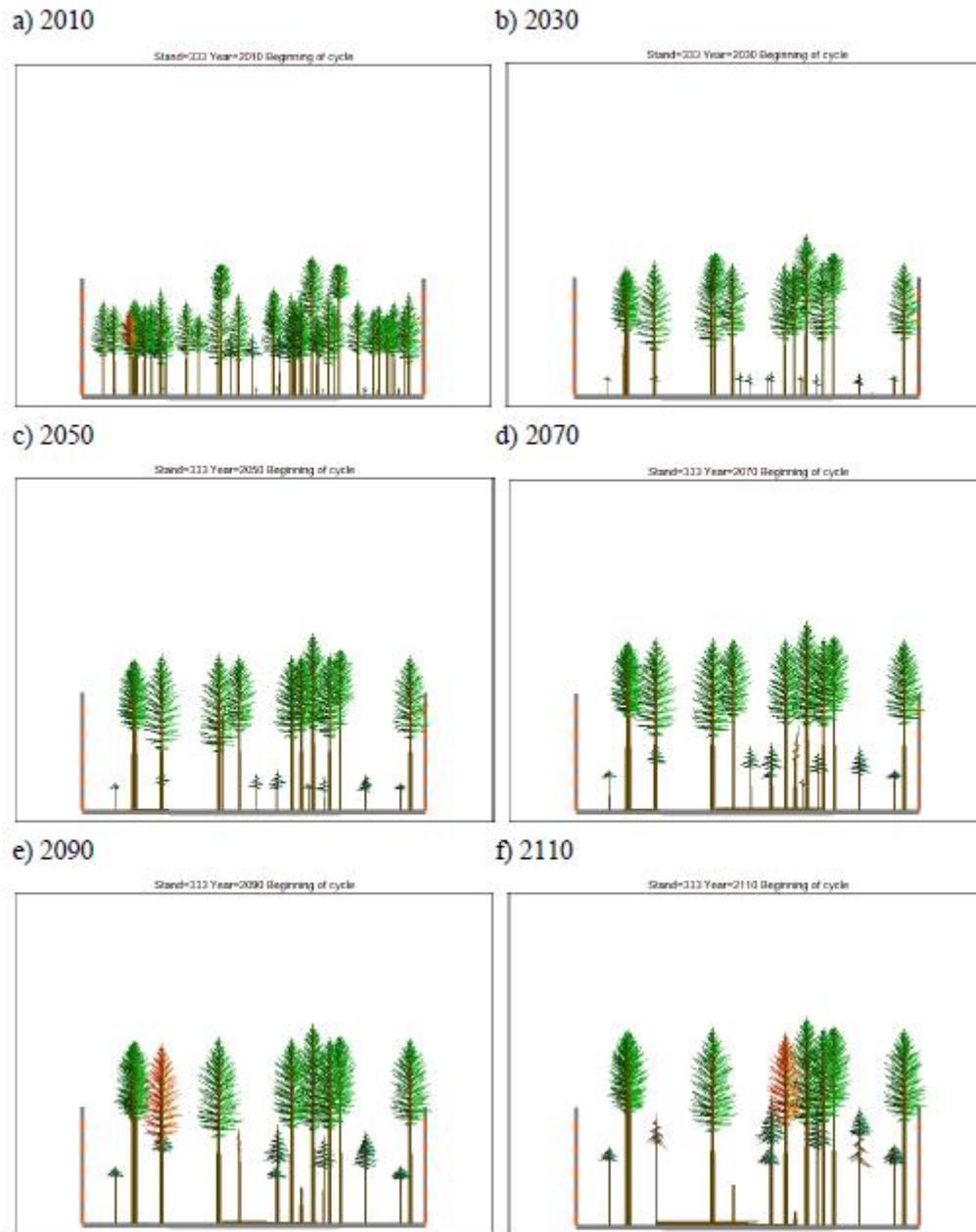


Figure A1-7. SVS profiles of the heavy thinning scenario in the dry moderate density Freda stand.

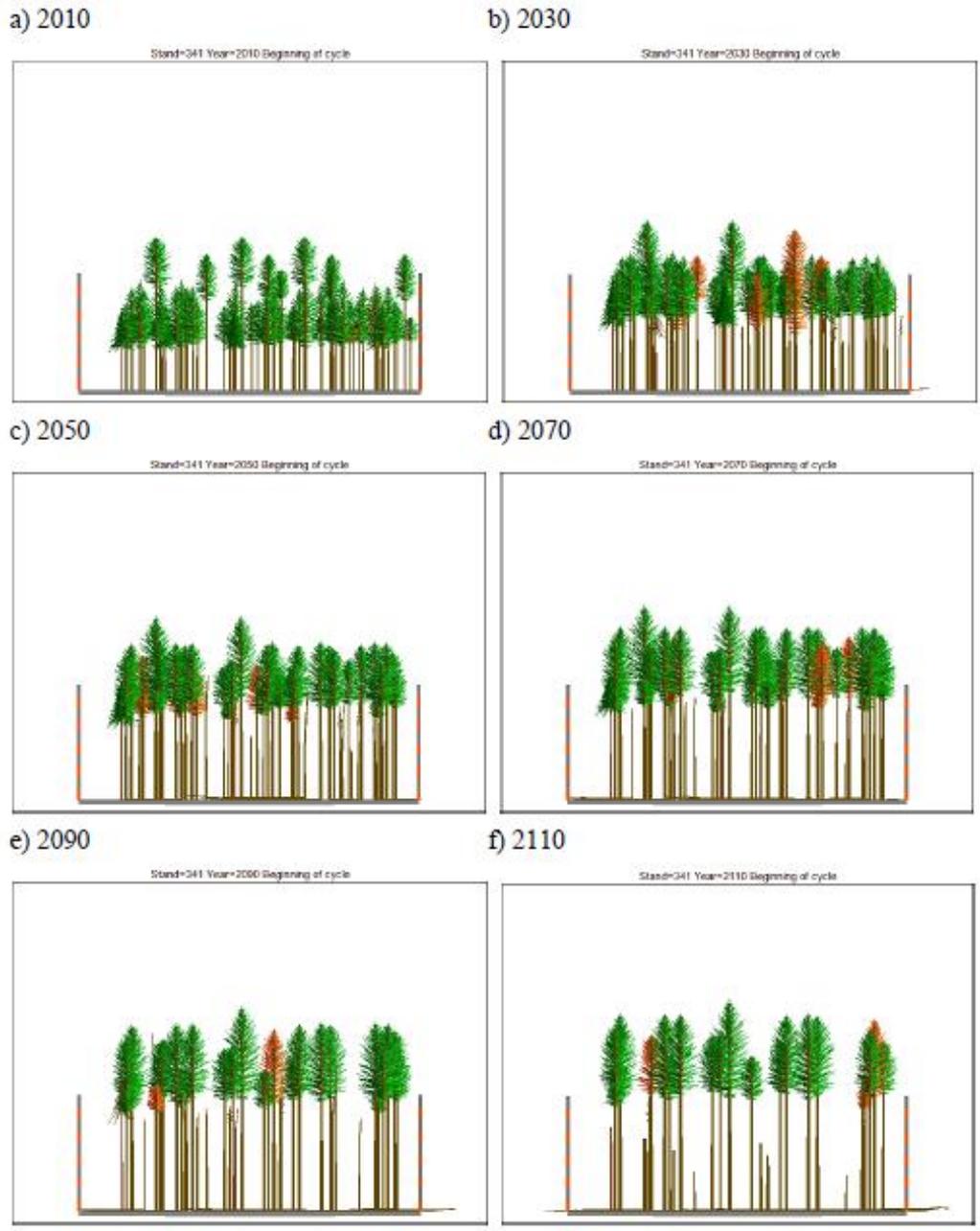


Figure A1-8. SVS profiles of the no treatment scenario in the dry high density Freda stand.

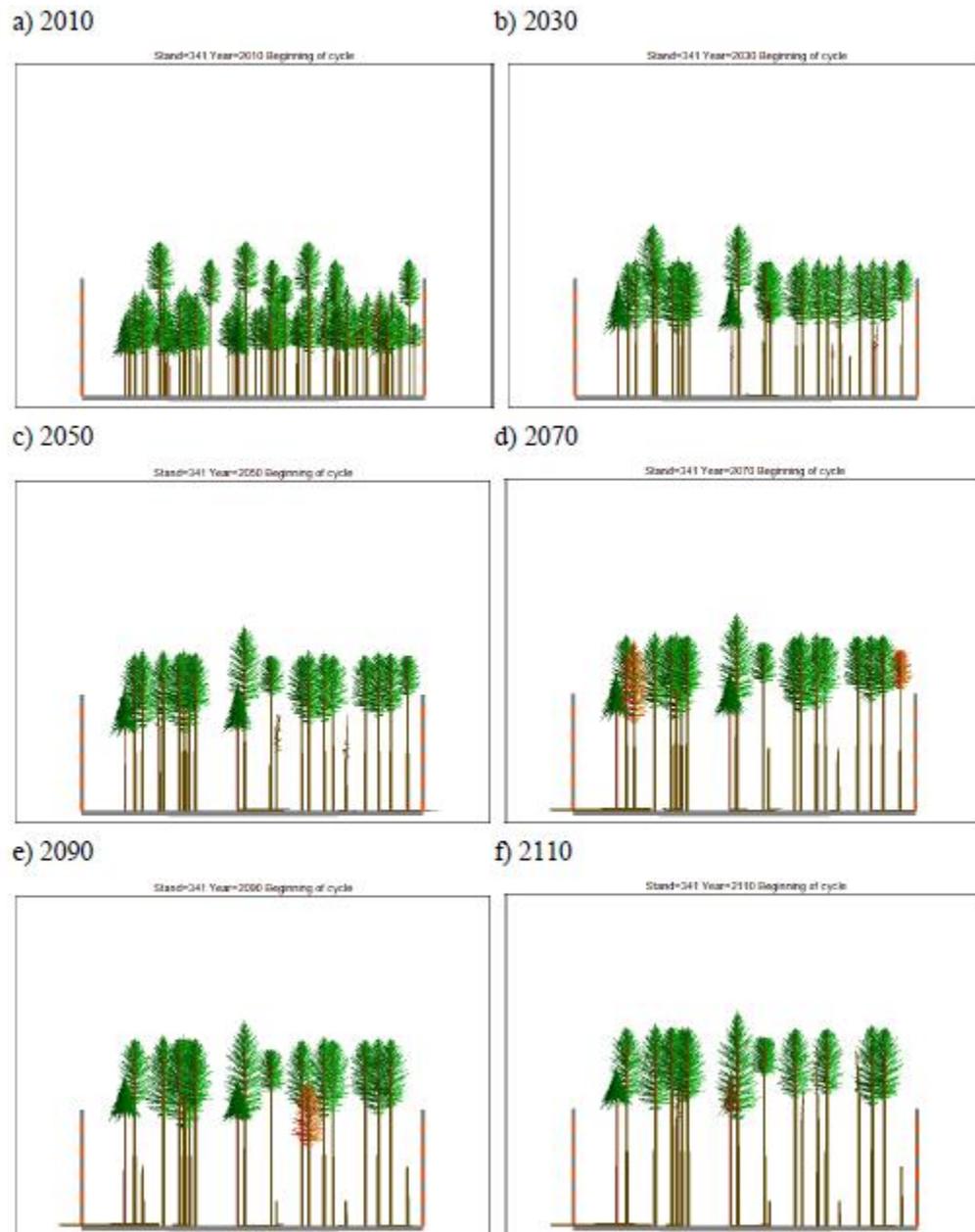


Figure A1-9. SVS profiles of the planned thinning scenario in the dry high density Freda stand.

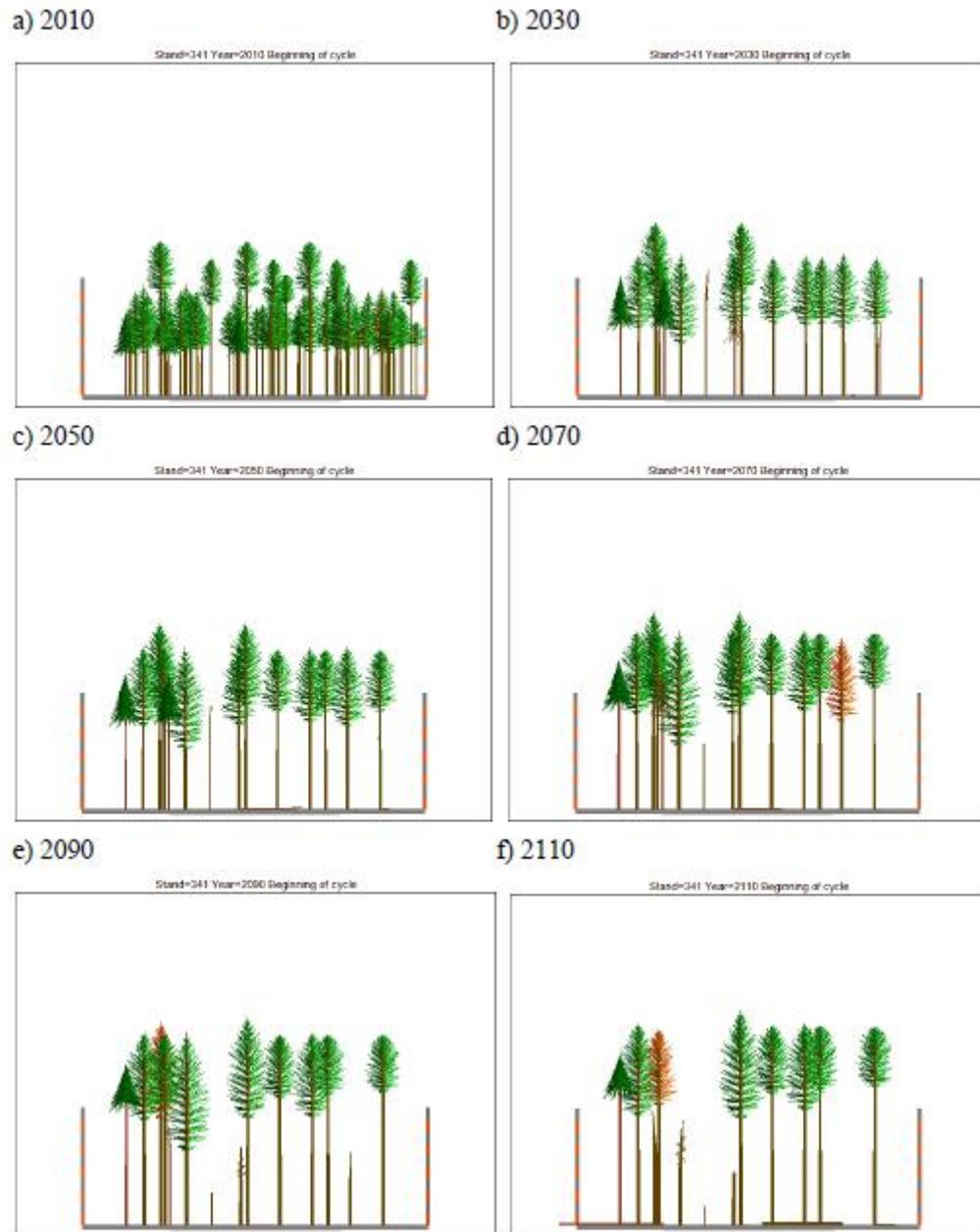


Figure A1-10. SVS profiles of the heavy thinning scenario in the dry high density Freda stand.

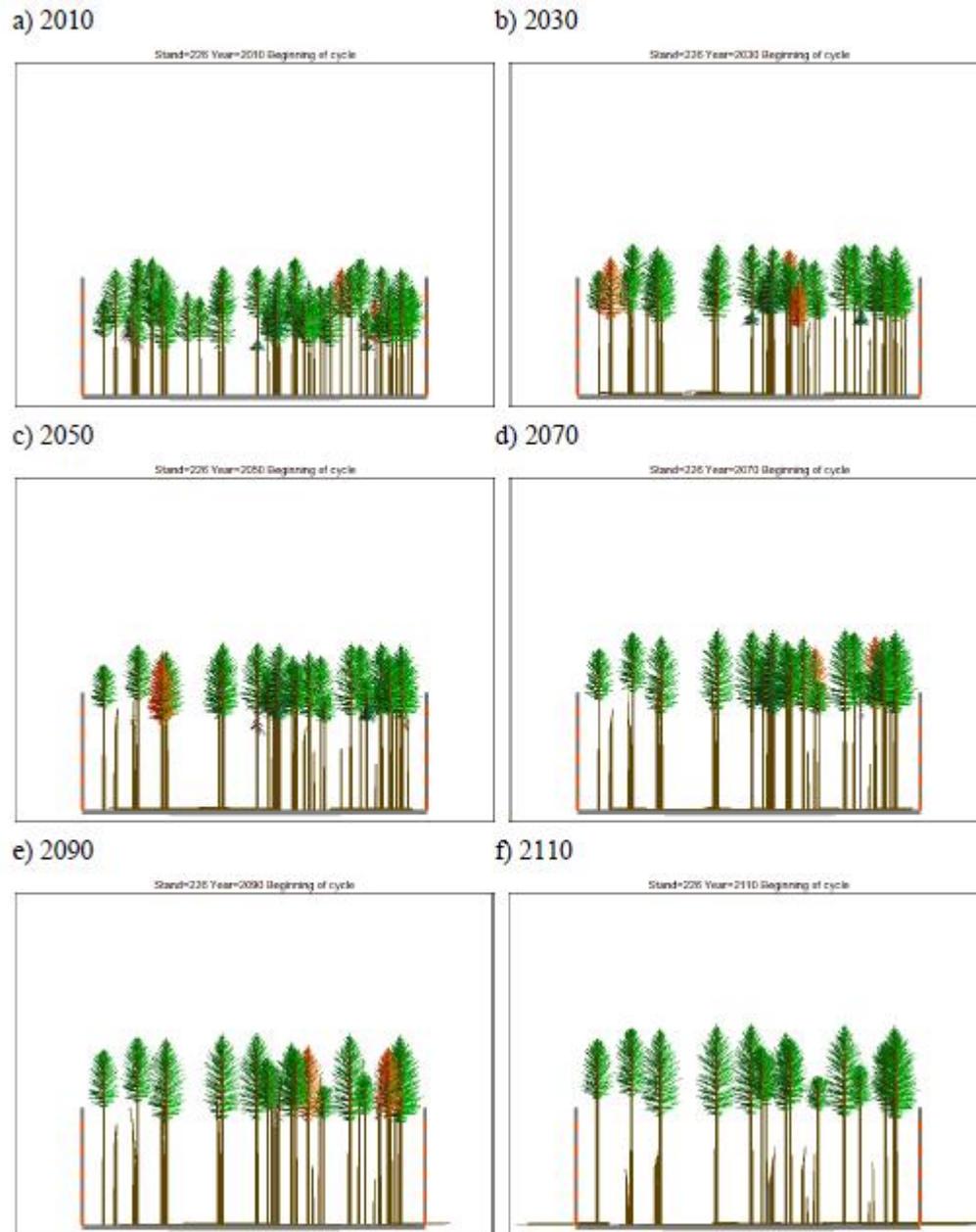


Figure A1-11. SVS profiles of the no treatment scenario in the wet moderate density Freda stand.

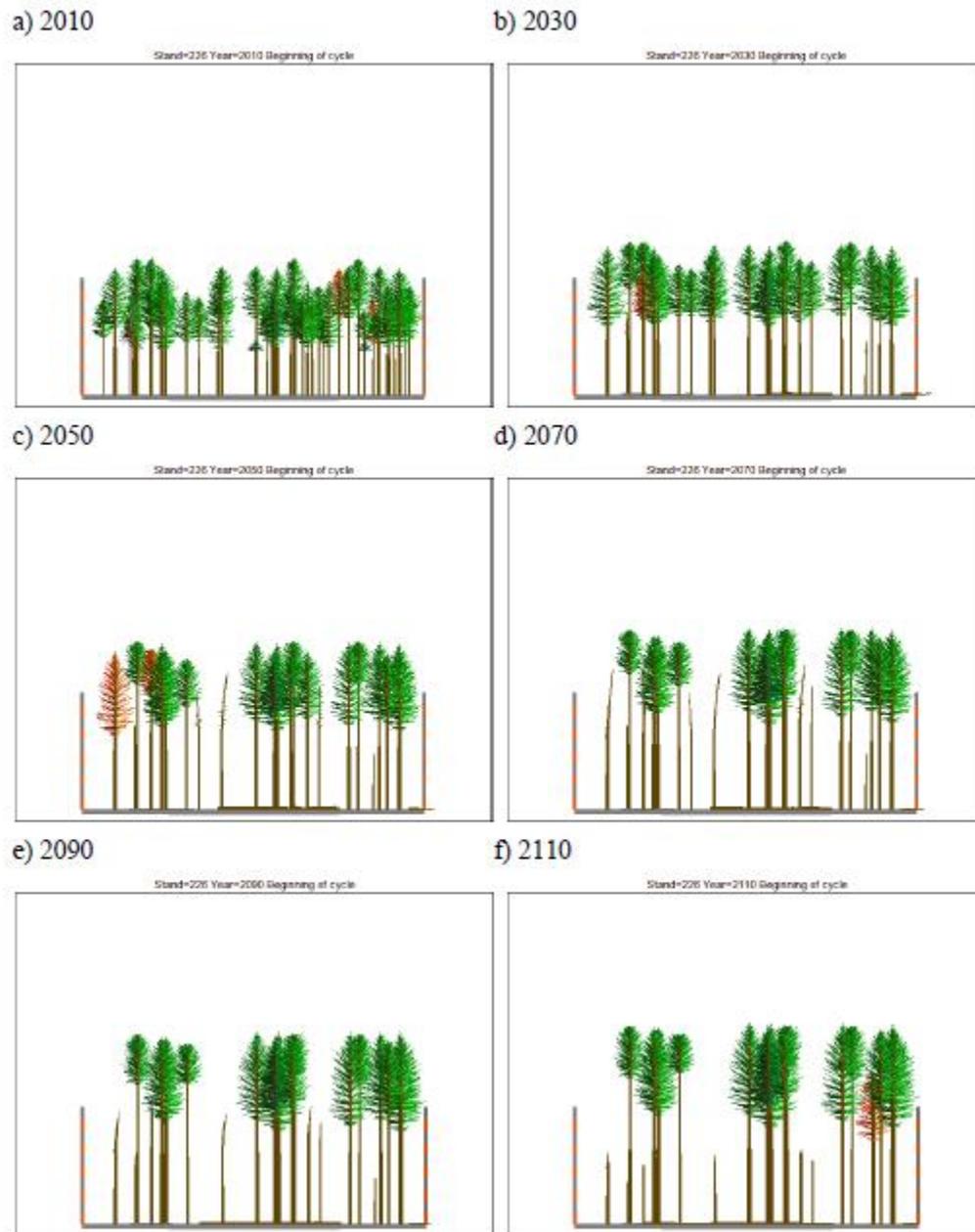


Figure A1-12. SVS profiles of the planned treatment scenario in the wet moderate density Freda stand.

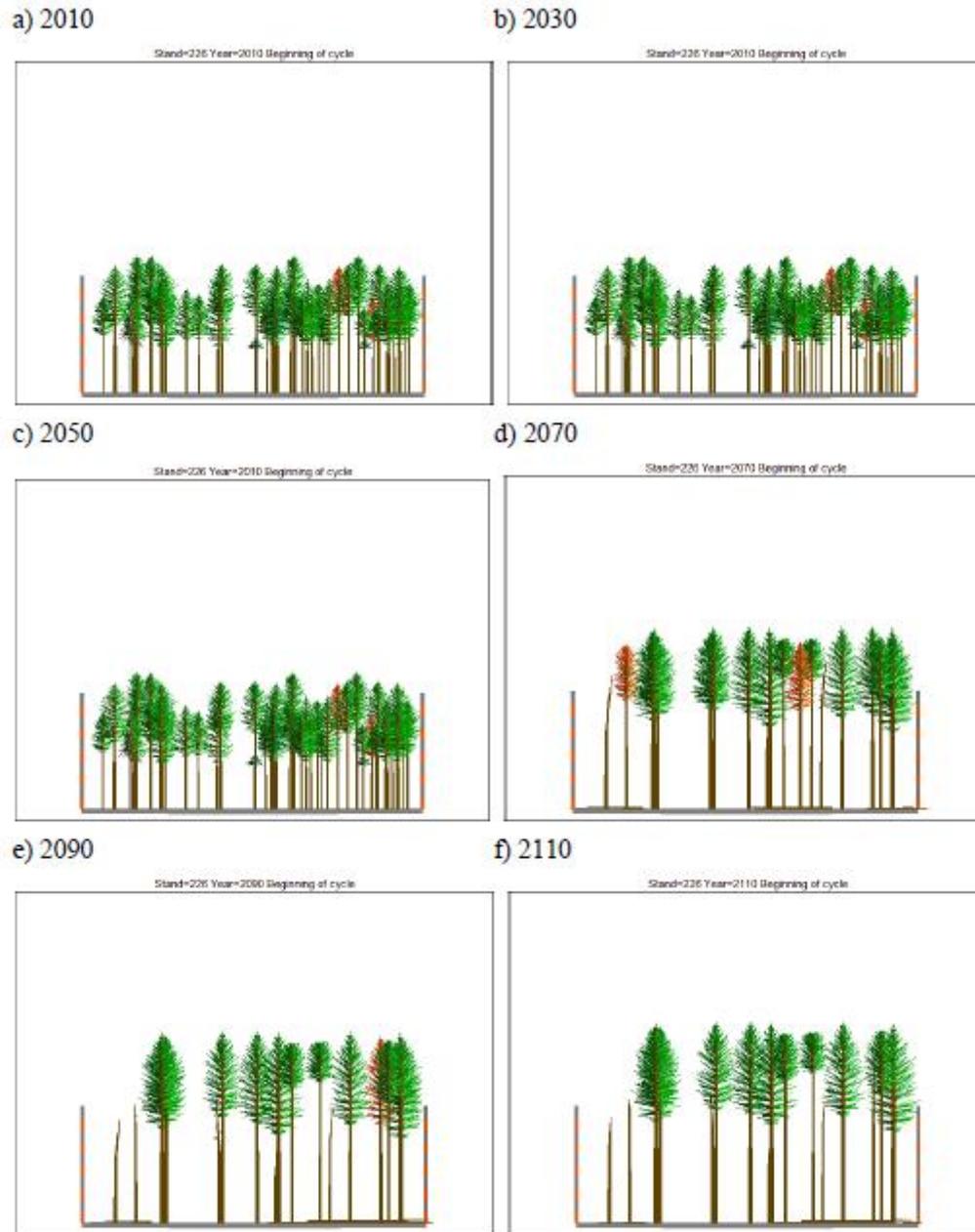


Figure A1-13. SVS profiles of the heavy treatment scenario in the wet moderate density Freda stand.

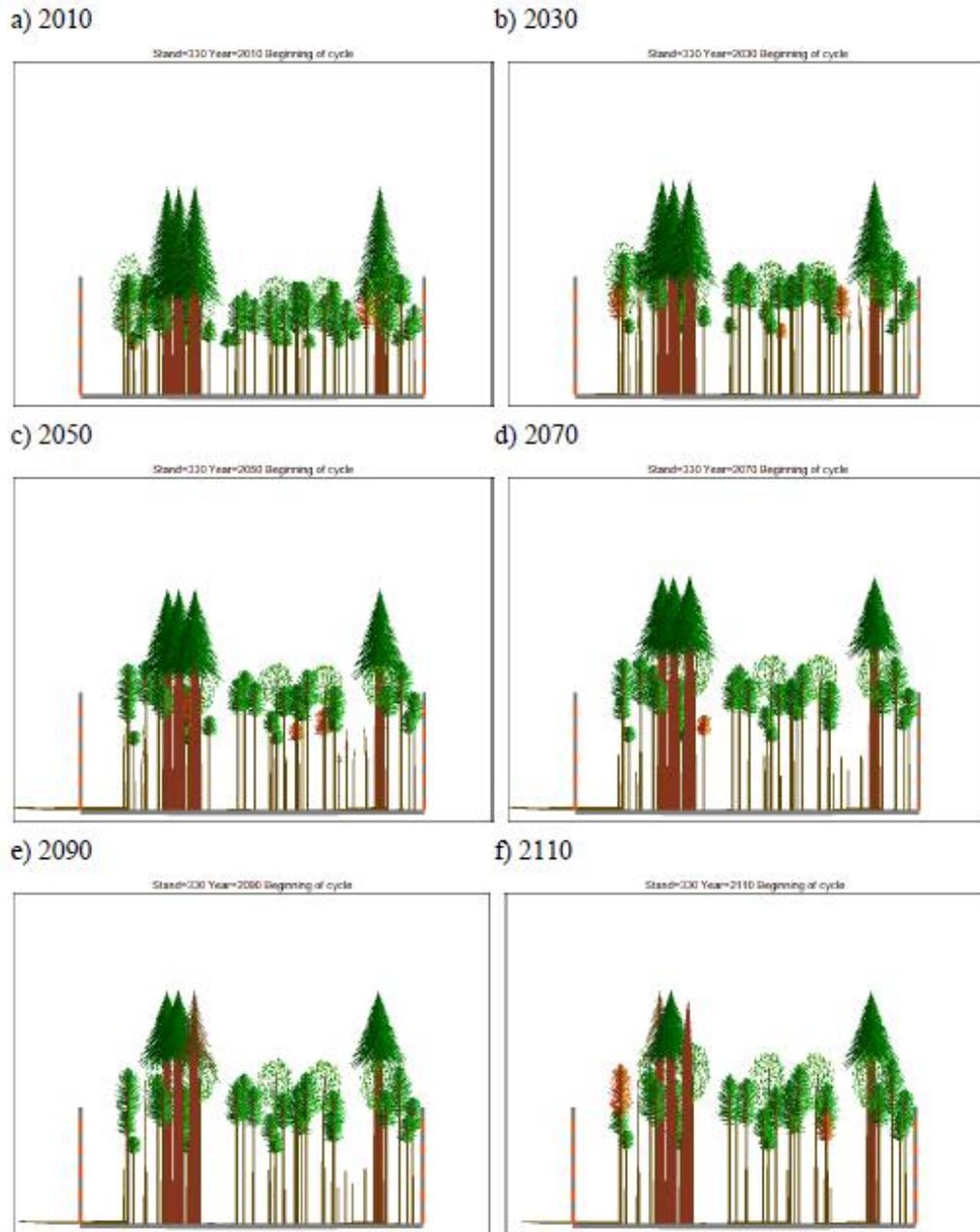


Figure A1-14. SVS profiles of the no treatment scenario in the wet high density Freda stand.

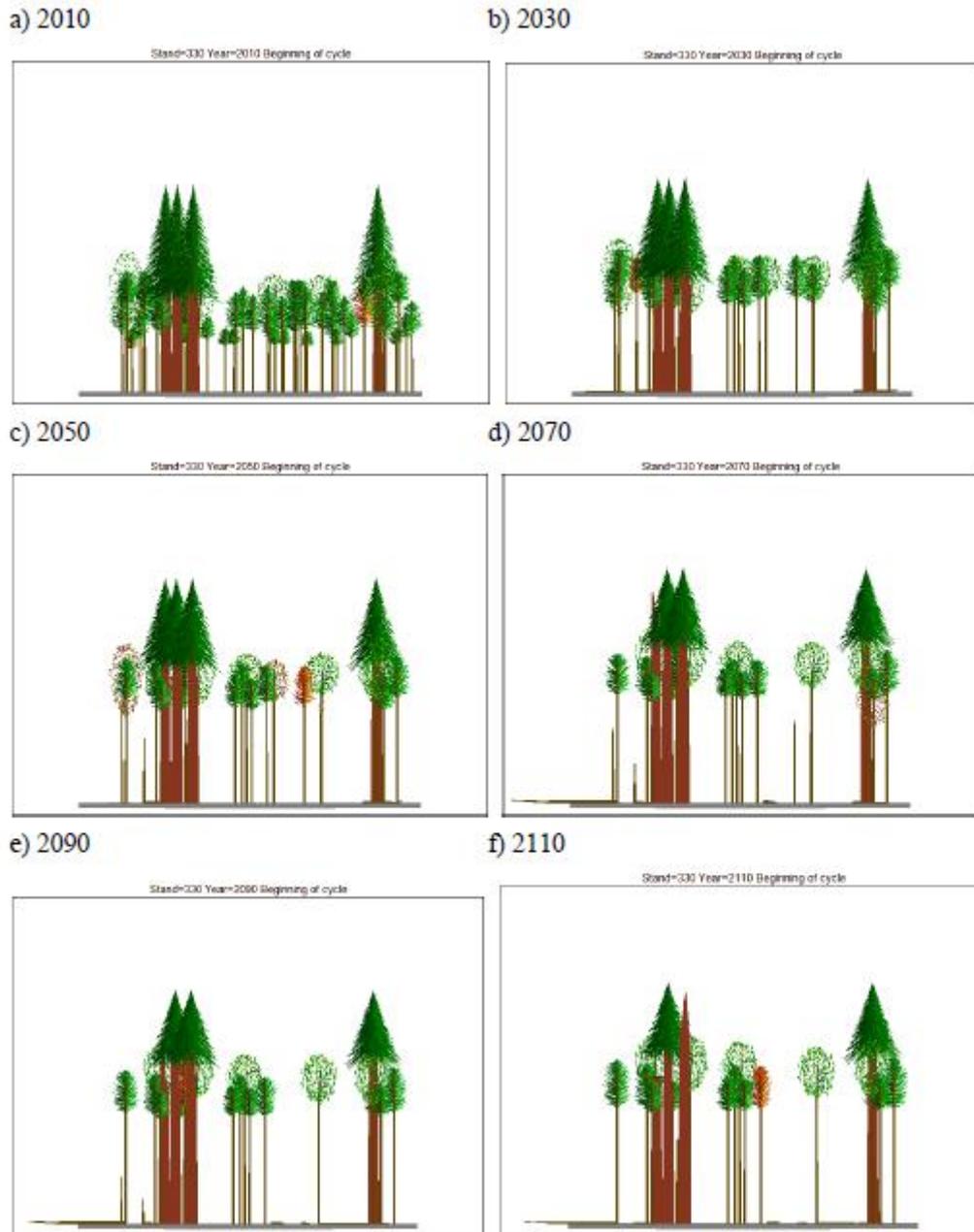


Figure A1-15. SVS profiles of the planned thinning scenario in the wet high density Freda stand.

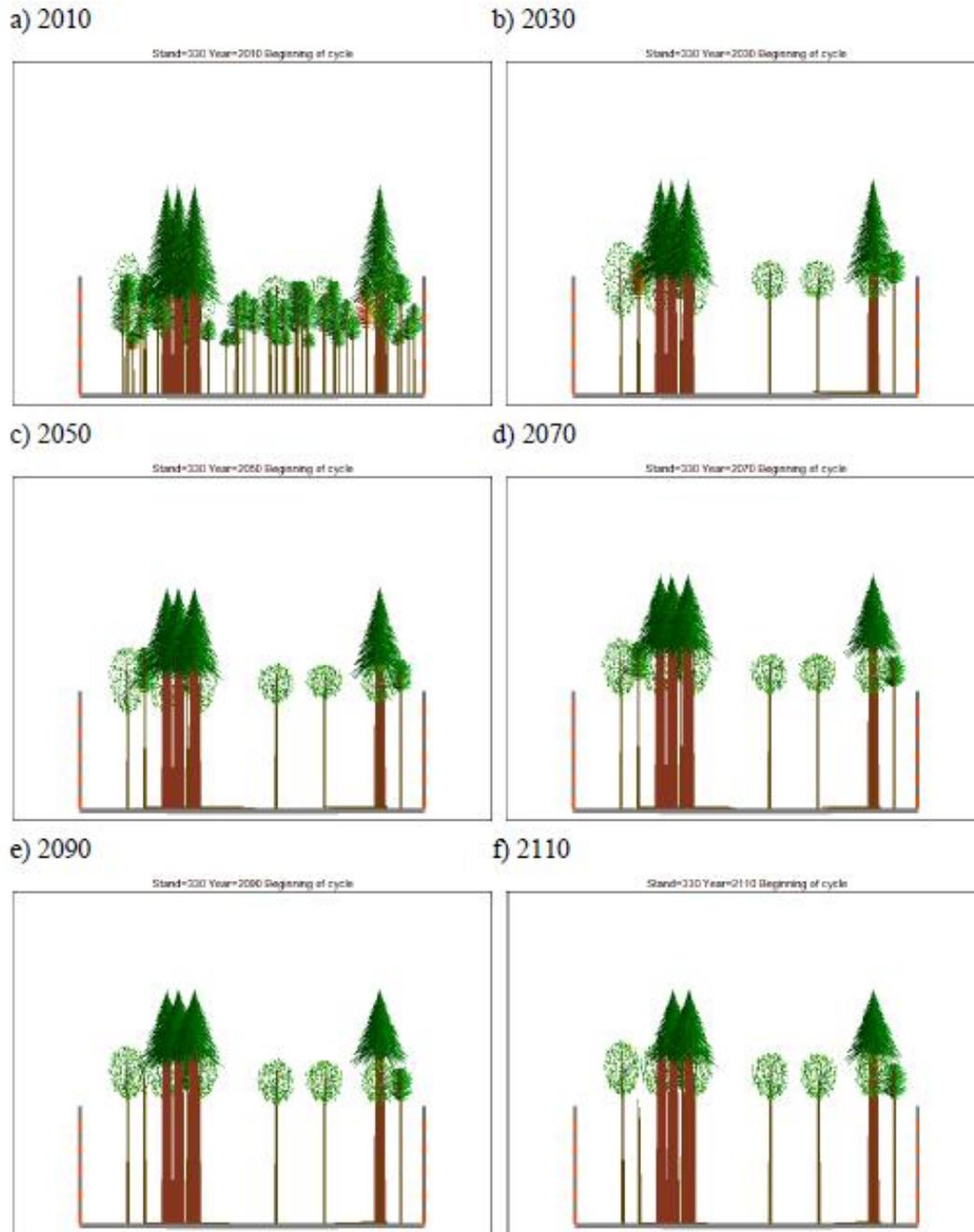


Figure A1-16. SVS profiles of the heavy thinning scenario in the wet high density Freda stand.

Appendix 2 – SVS profiles of 2006 windthrow stands

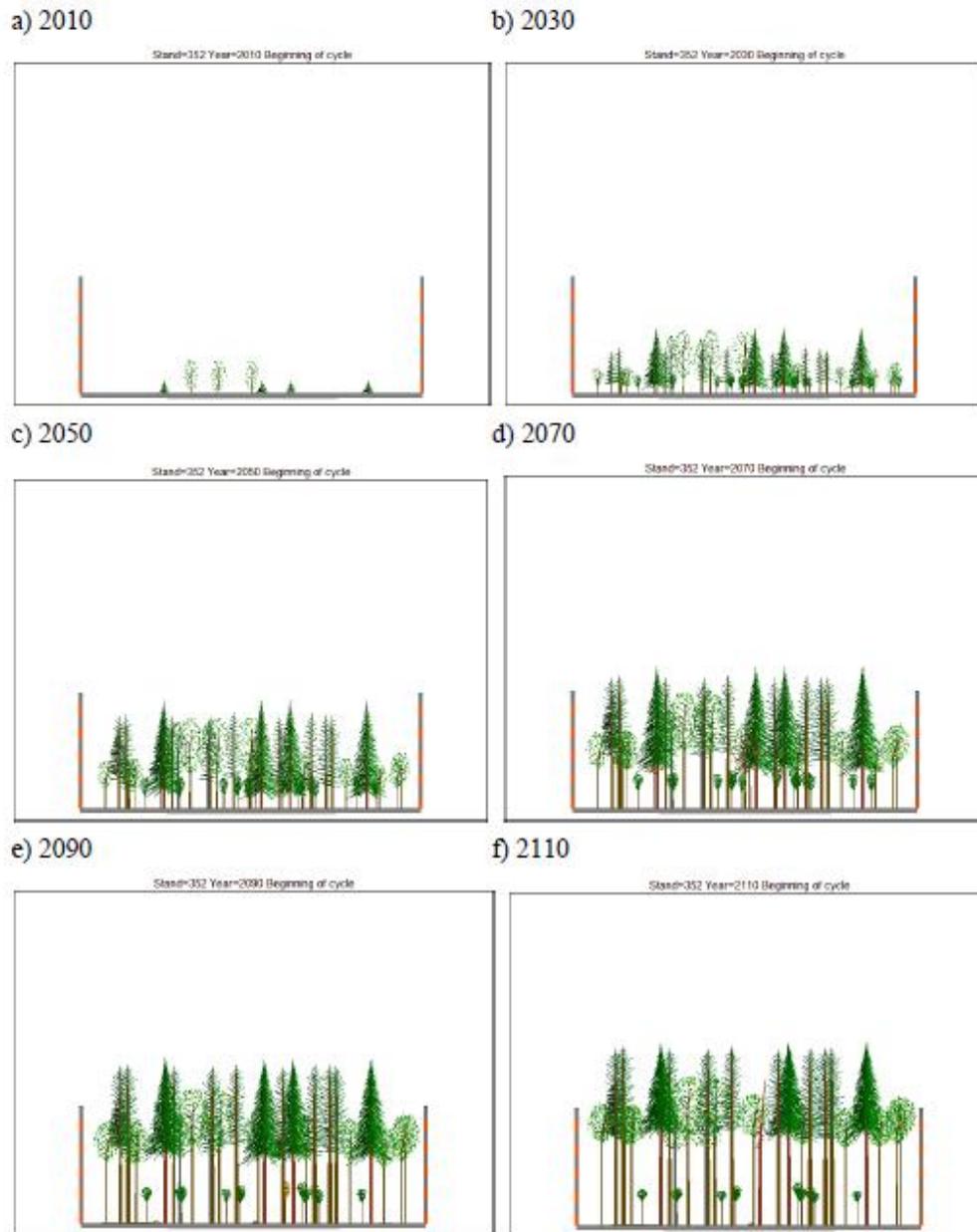


Figure A2-1. SVS profiles of the no treatment scenario in the dry low residual 2006 disturbance stand.

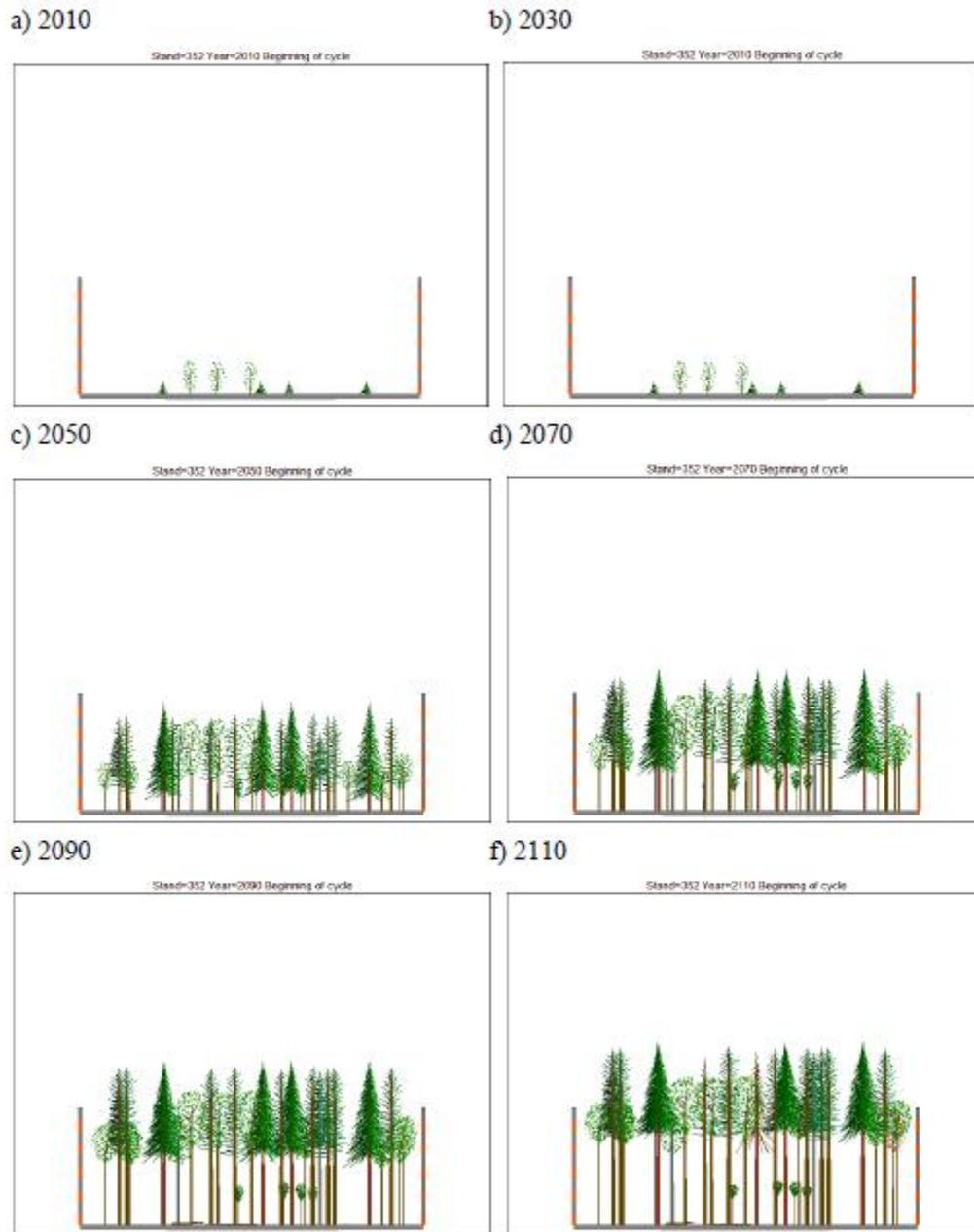


Figure A2-2. SVS profiles of the light thinning scenario in the dry low residual 2006 disturbance stand.

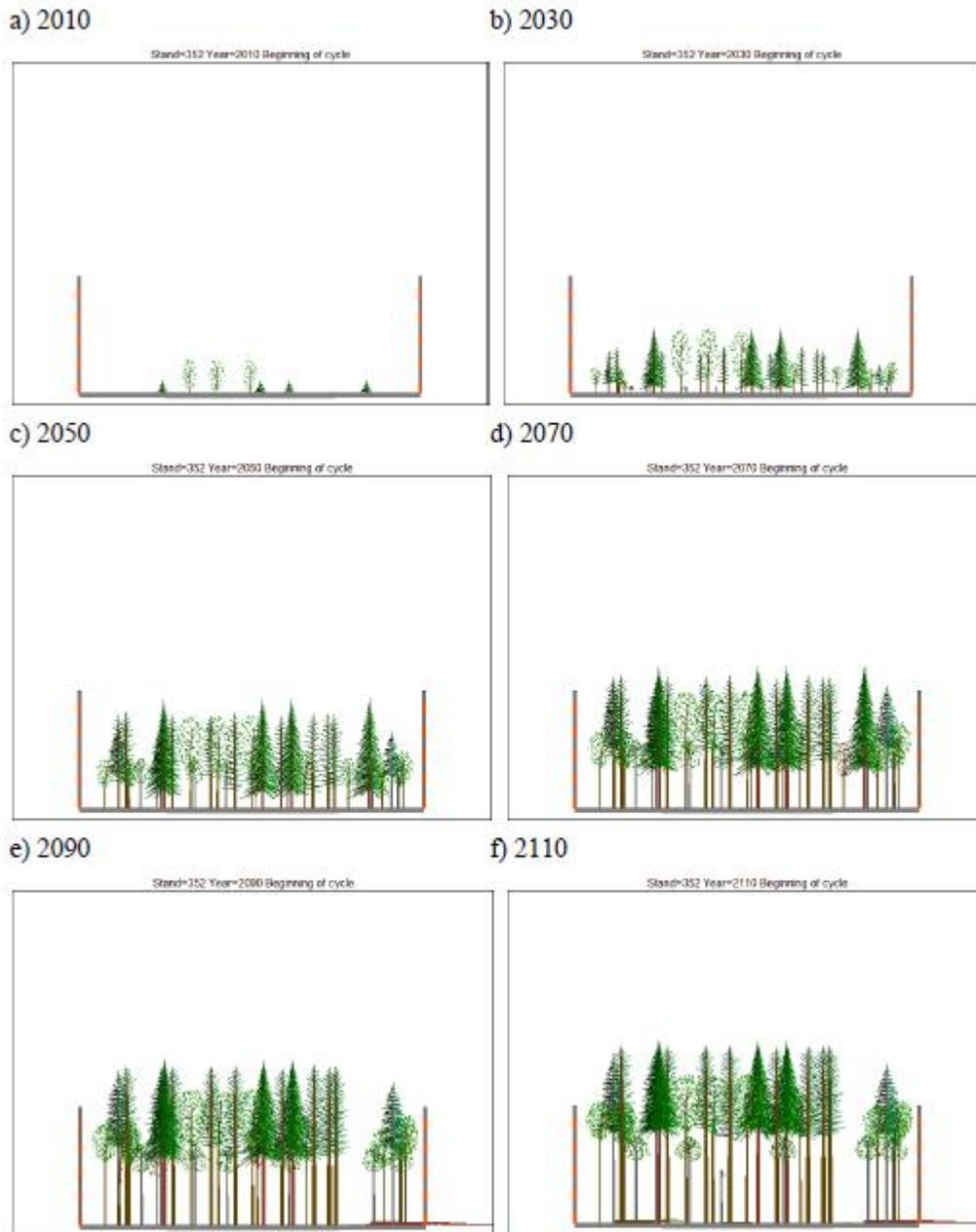


Figure A2-3. SVS profiles of the heavy thinning scenario in the dry low residual 2006 disturbance stand.

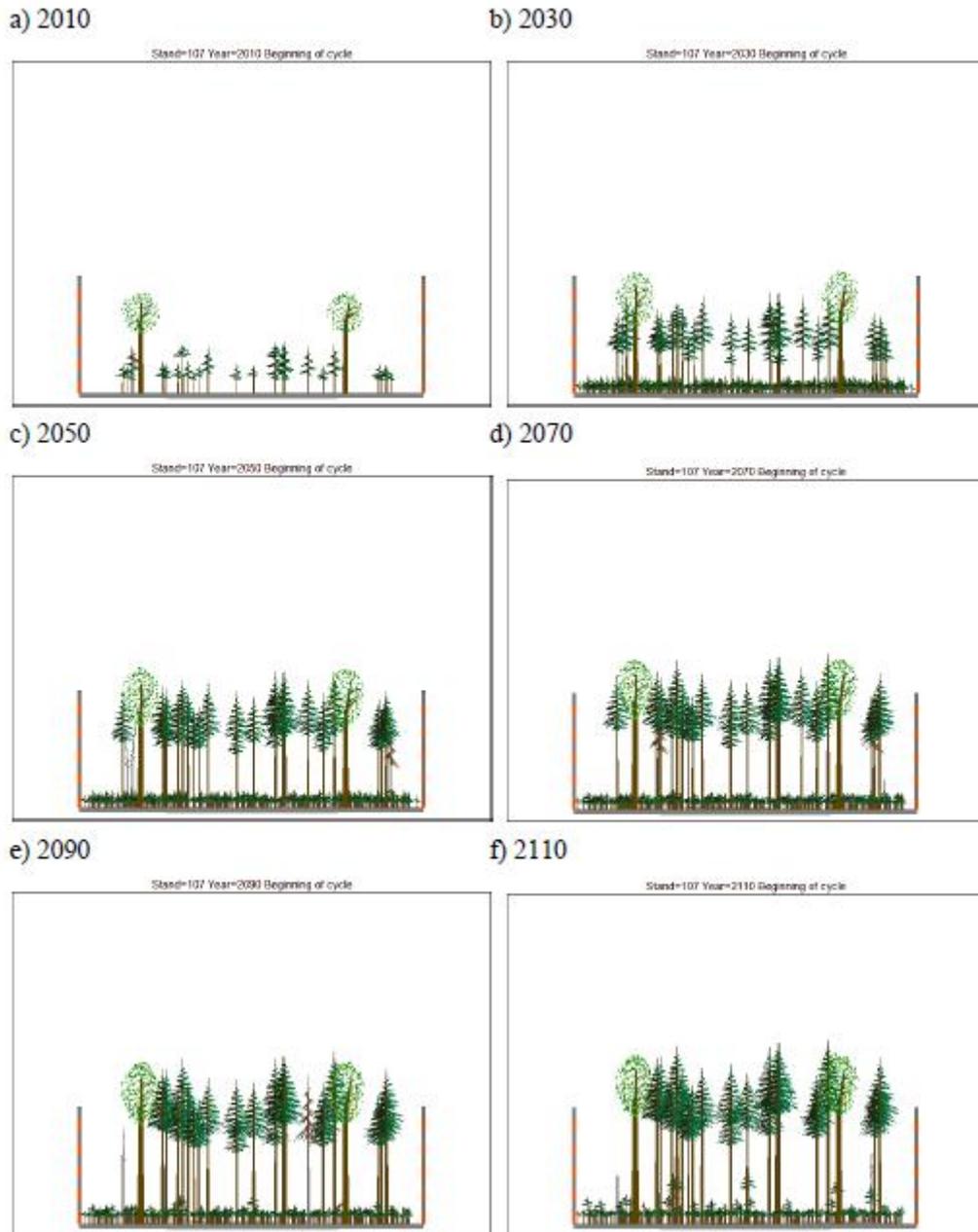


Figure A2-4. SVS profiles of the no treatment scenario in the dry medium residual2006 disturbance stand.

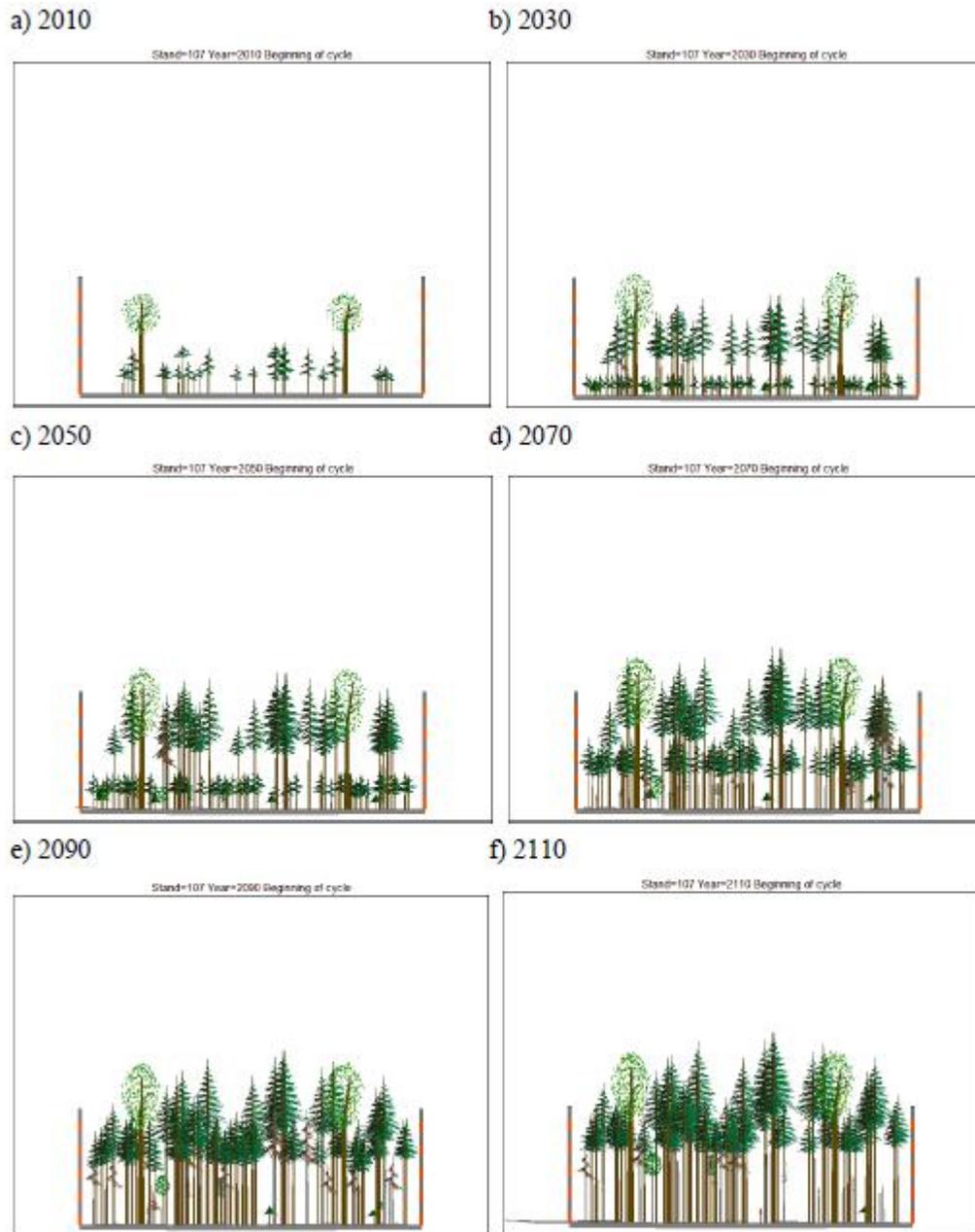


Figure A2-5. SVS profiles of the light thinning scenario in the dry medium residual 2006 disturbance stand.

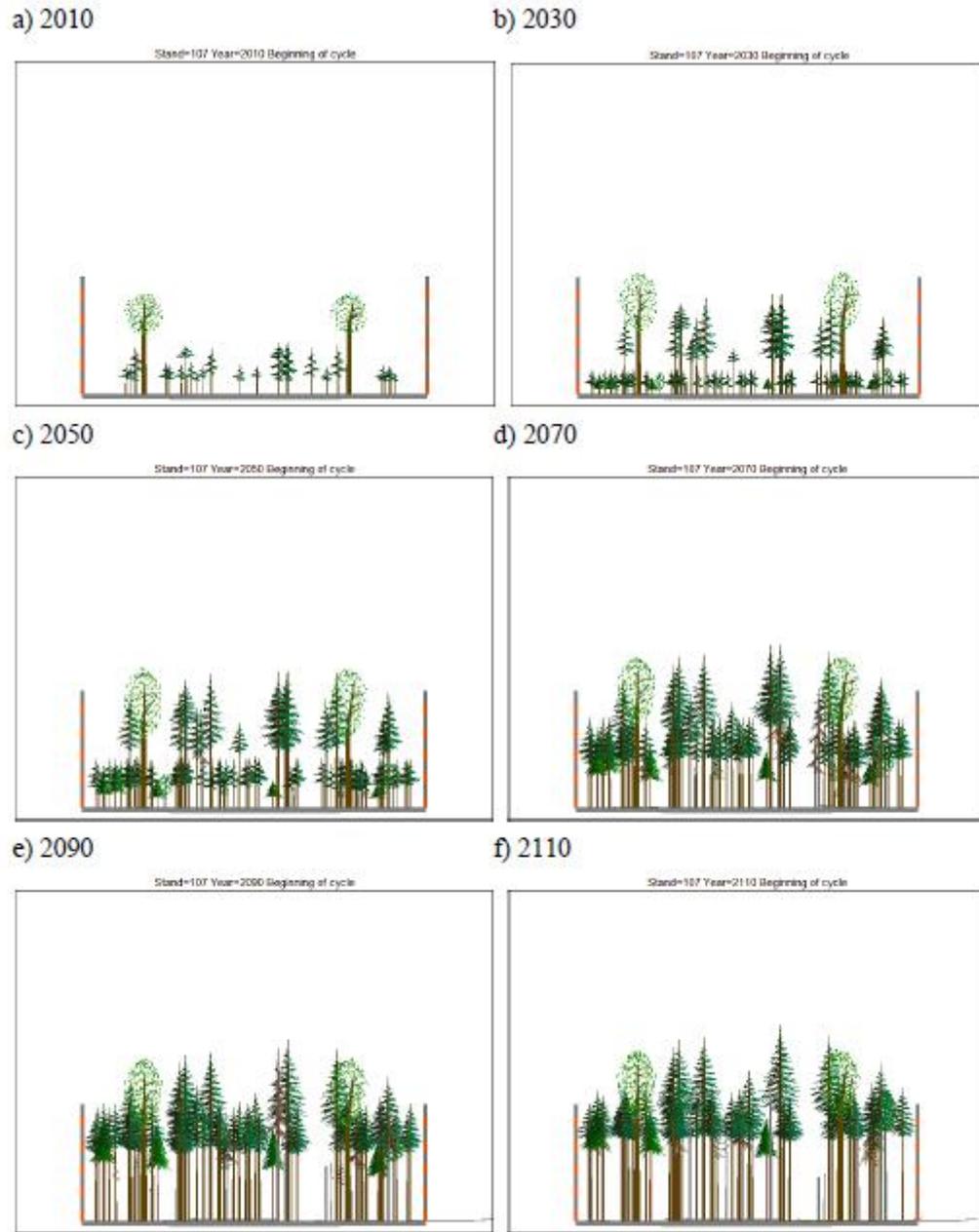


Figure A2-6. SVS profiles of the moderate thinning scenario in the dry medium residual 2006 disturbance stand.

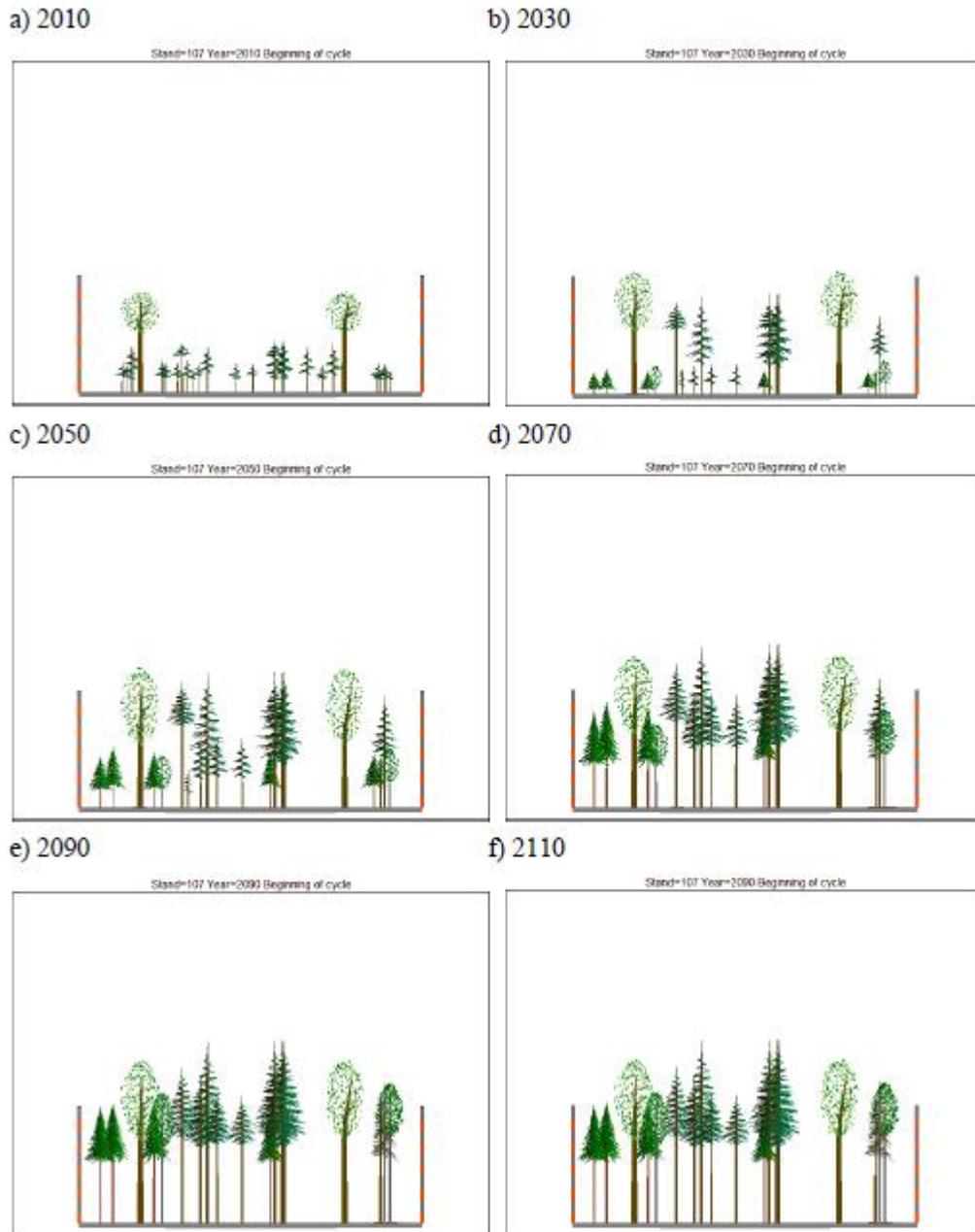


Figure A2-7. SVS profiles of the heavy thinning scenario in the dry medium residual 2006 disturbance stand.

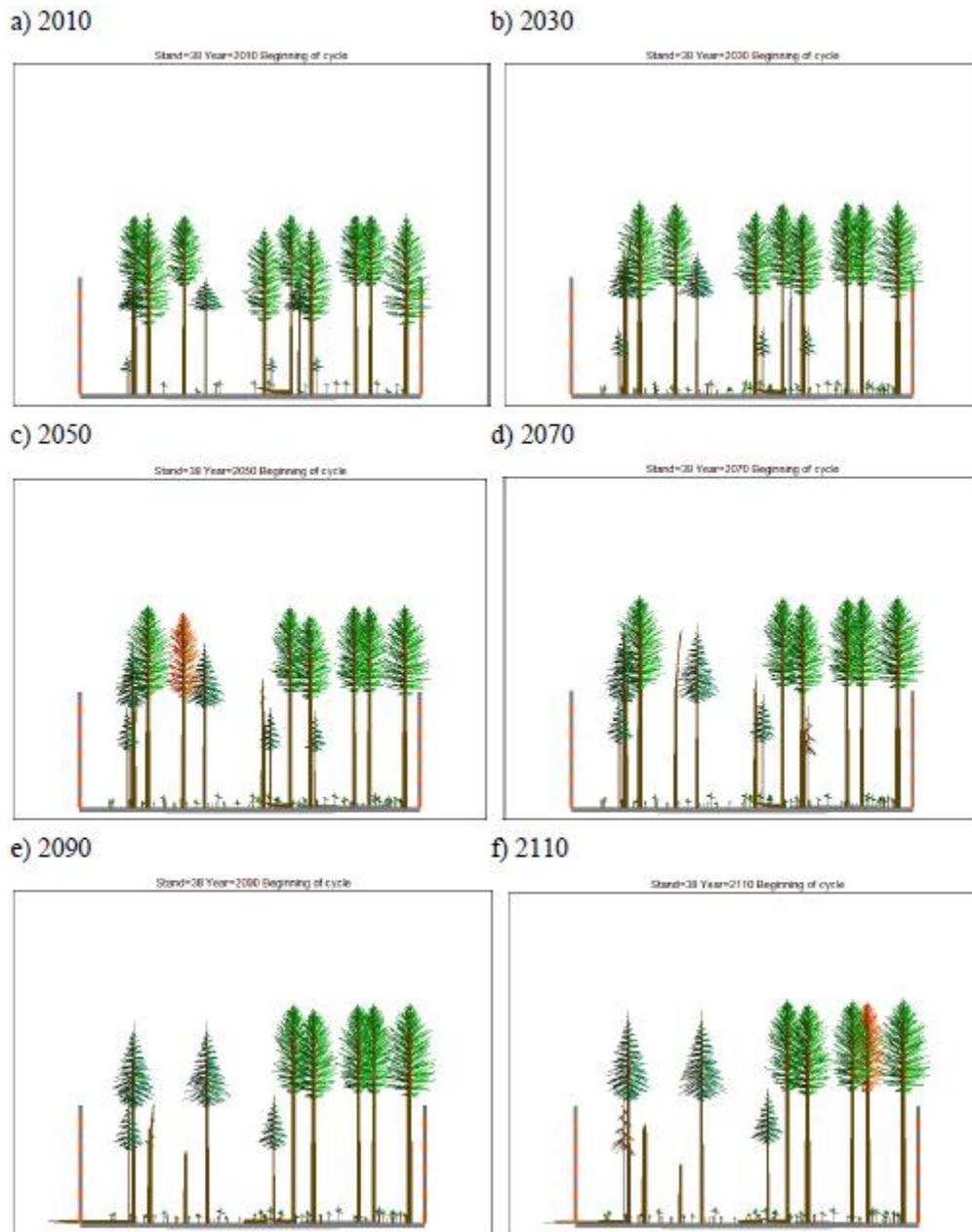


Figure A2-8. SVS profiles of the no treatment scenario in the dry high residual 2006 disturbance stand.

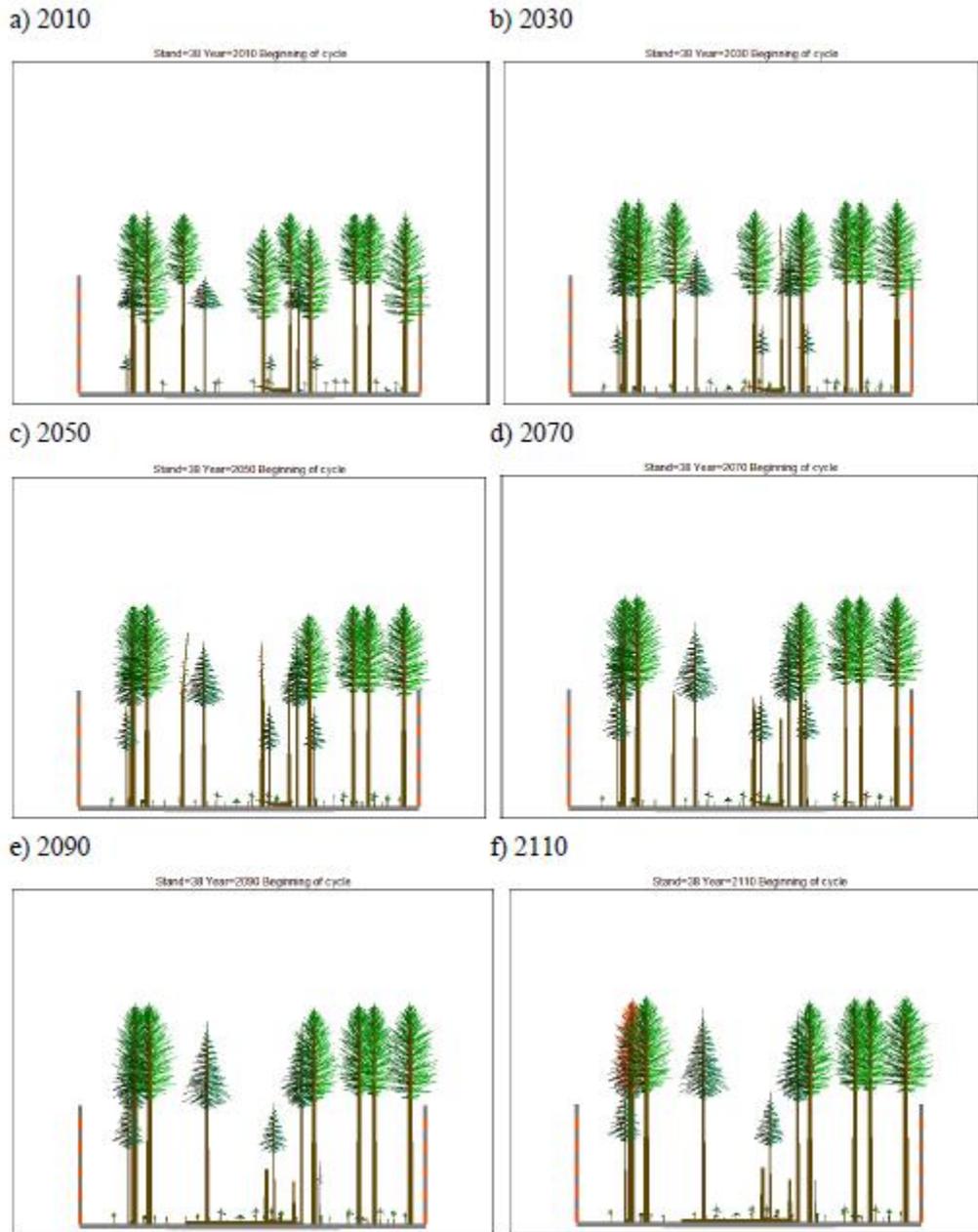


Figure A2-9. SVS profiles of the light thinning scenario in the dry high residual 2006 disturbance stand.

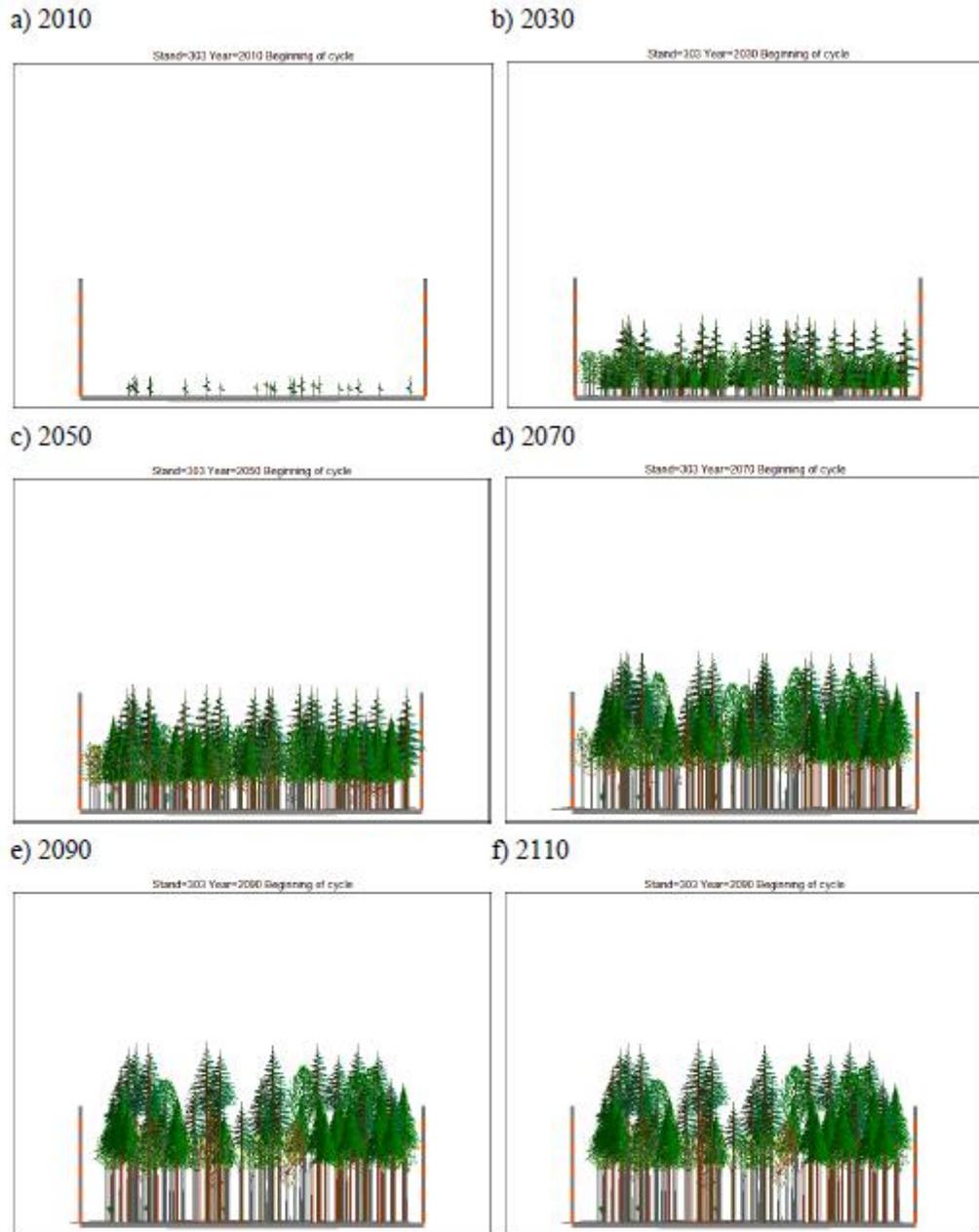


Figure A2-10. SVS profiles of the no treatment scenario in the wet low residual 2006 disturbance stand.

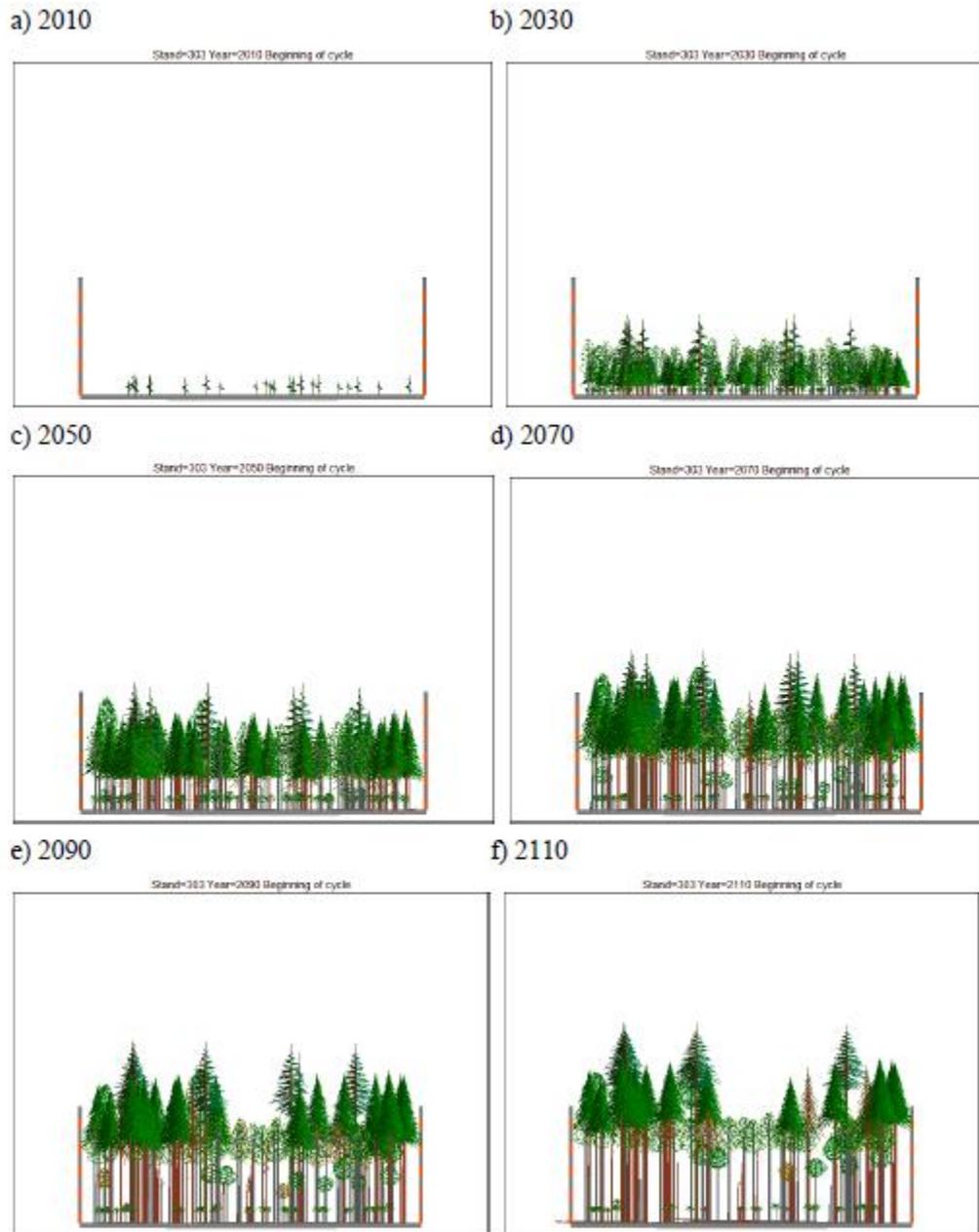


Figure A2-11. SVS profiles of the light thinning scenario in the wet low residual 2006 disturbance stand.

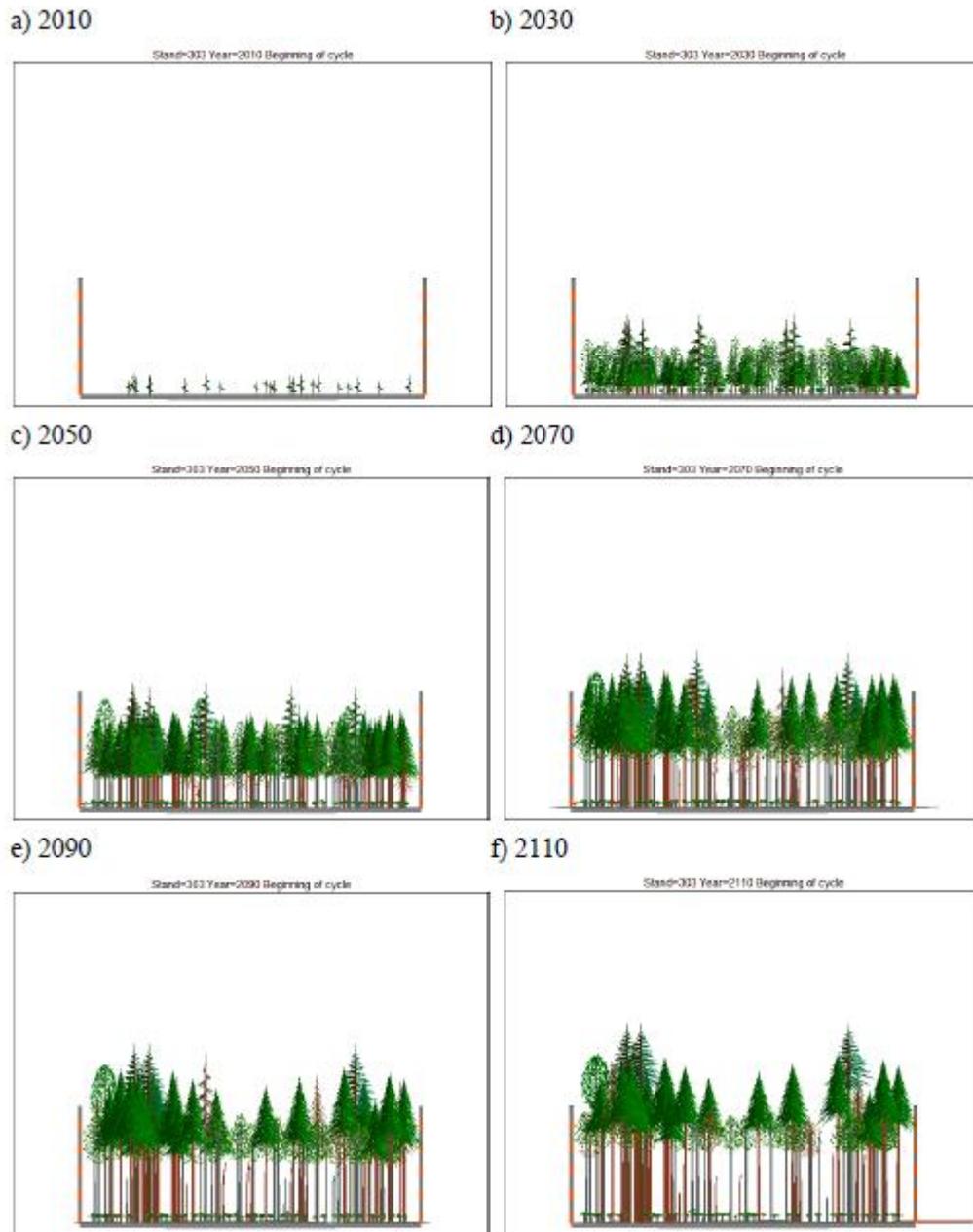


Figure A2-12. SVS profiles of the moderate thinning scenario in the wet low residual 2006 disturbance stand.

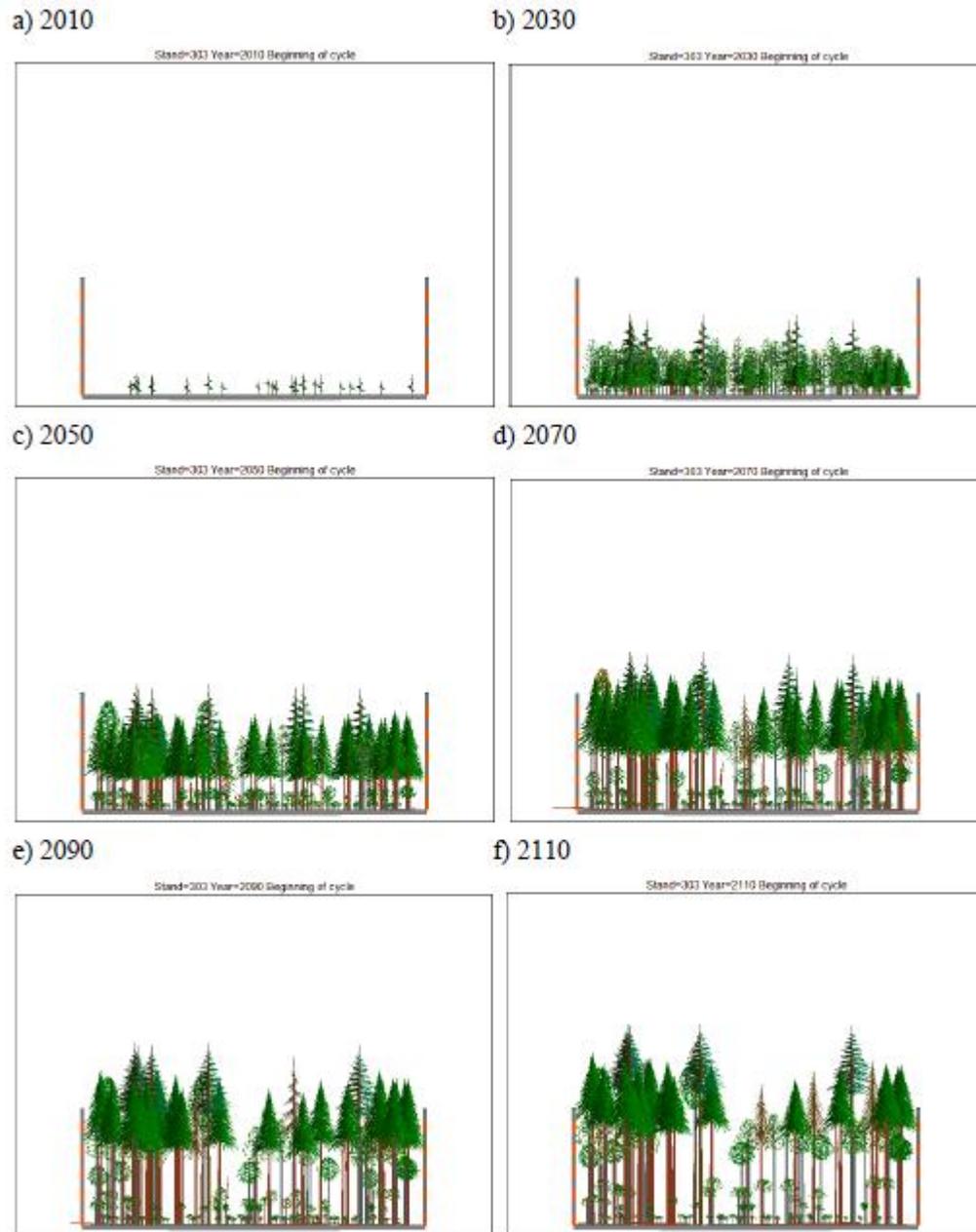


Figure A2-13. SVS profiles of the heavy thinning scenario in the wet low residual 2006 disturbance stand.

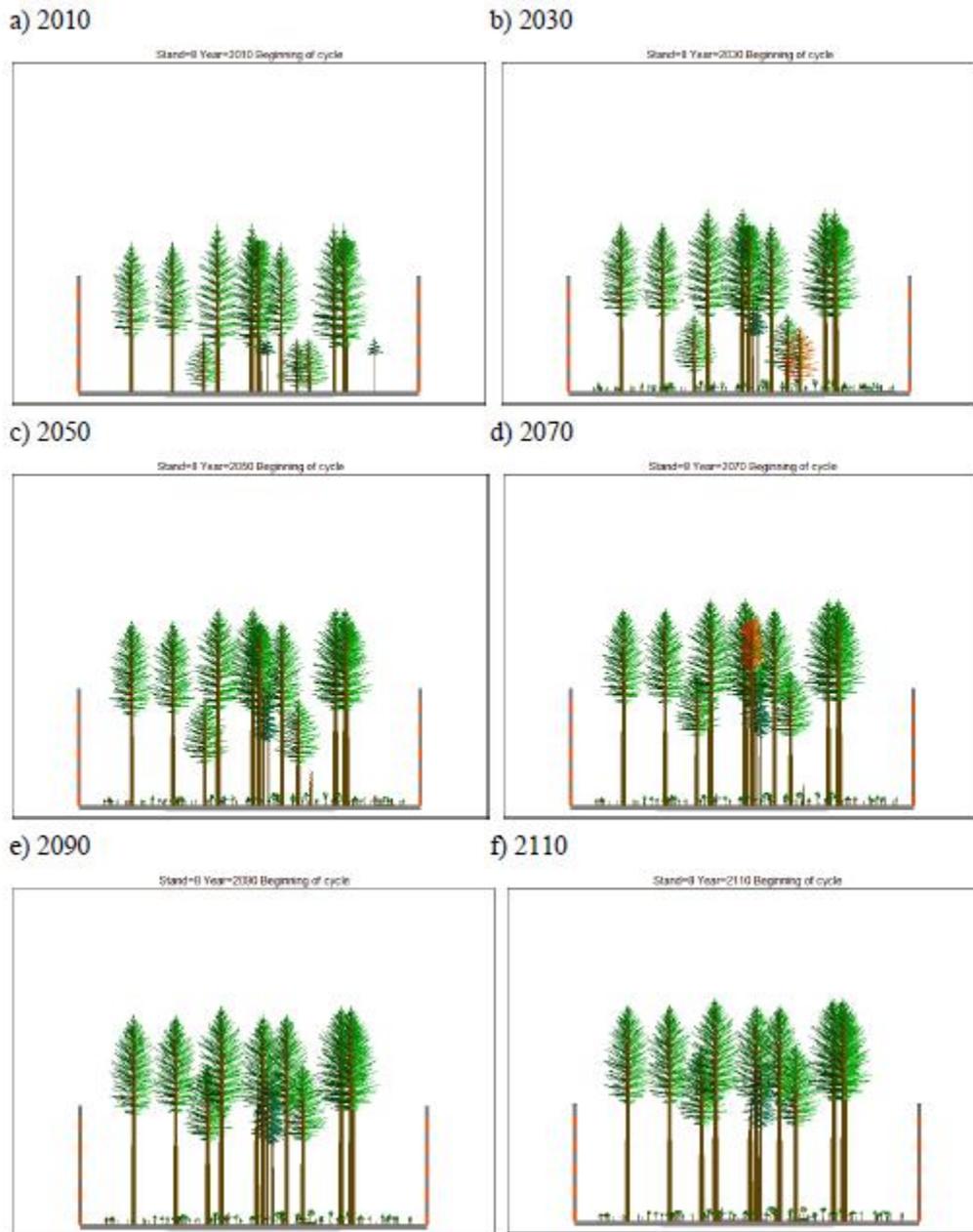


Figure A2-14.SVS profiles of the no treatment scenario in the wet medium residual 2006 disturbance stand.

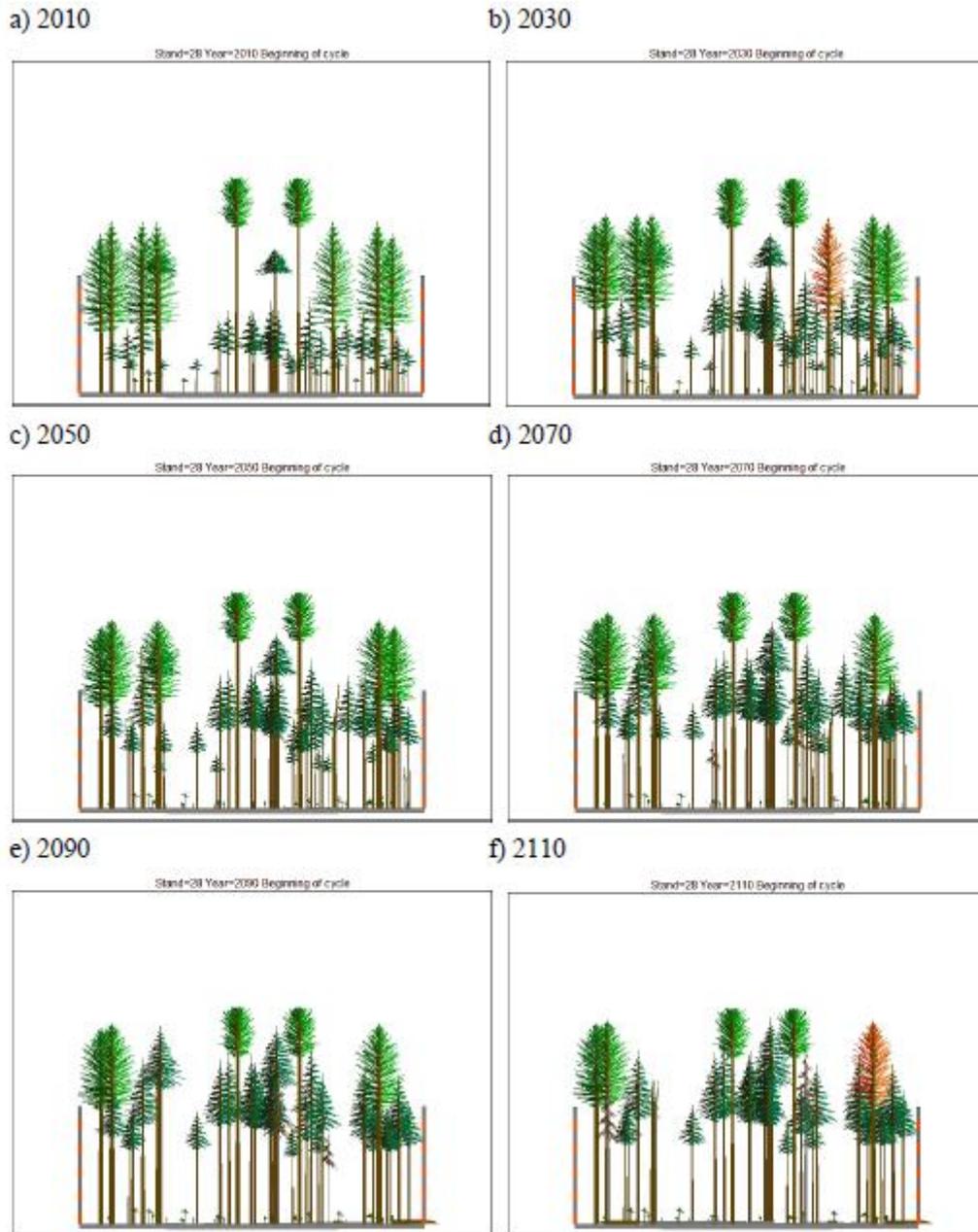


Figure A2-15. SVS profiles of the no treatment scenario in the wet high residual 2006 disturbance stand.

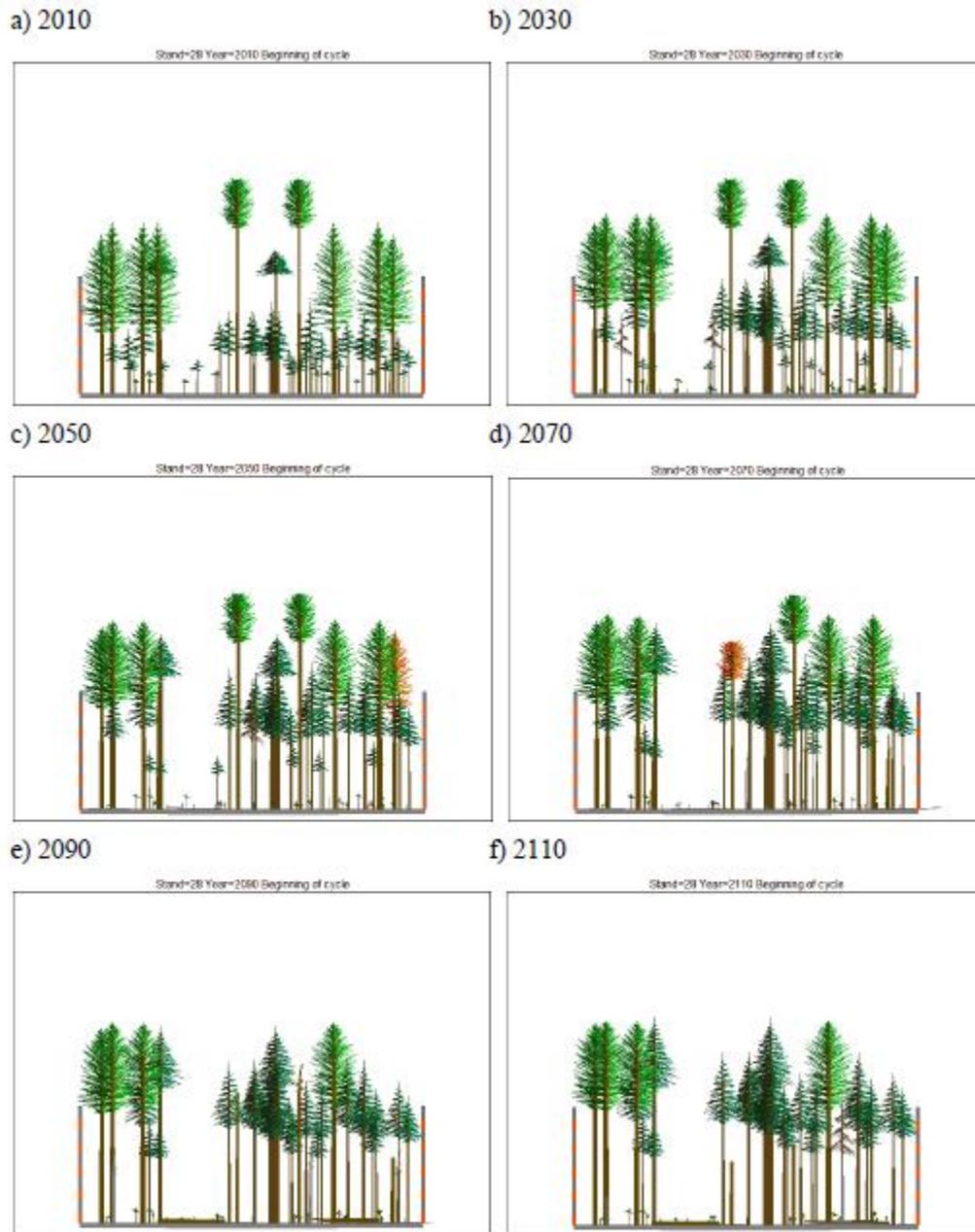


Figure A2-16. SVS profiles of the heavy thinning scenario in the wet high residual 2006 disturbance stand.

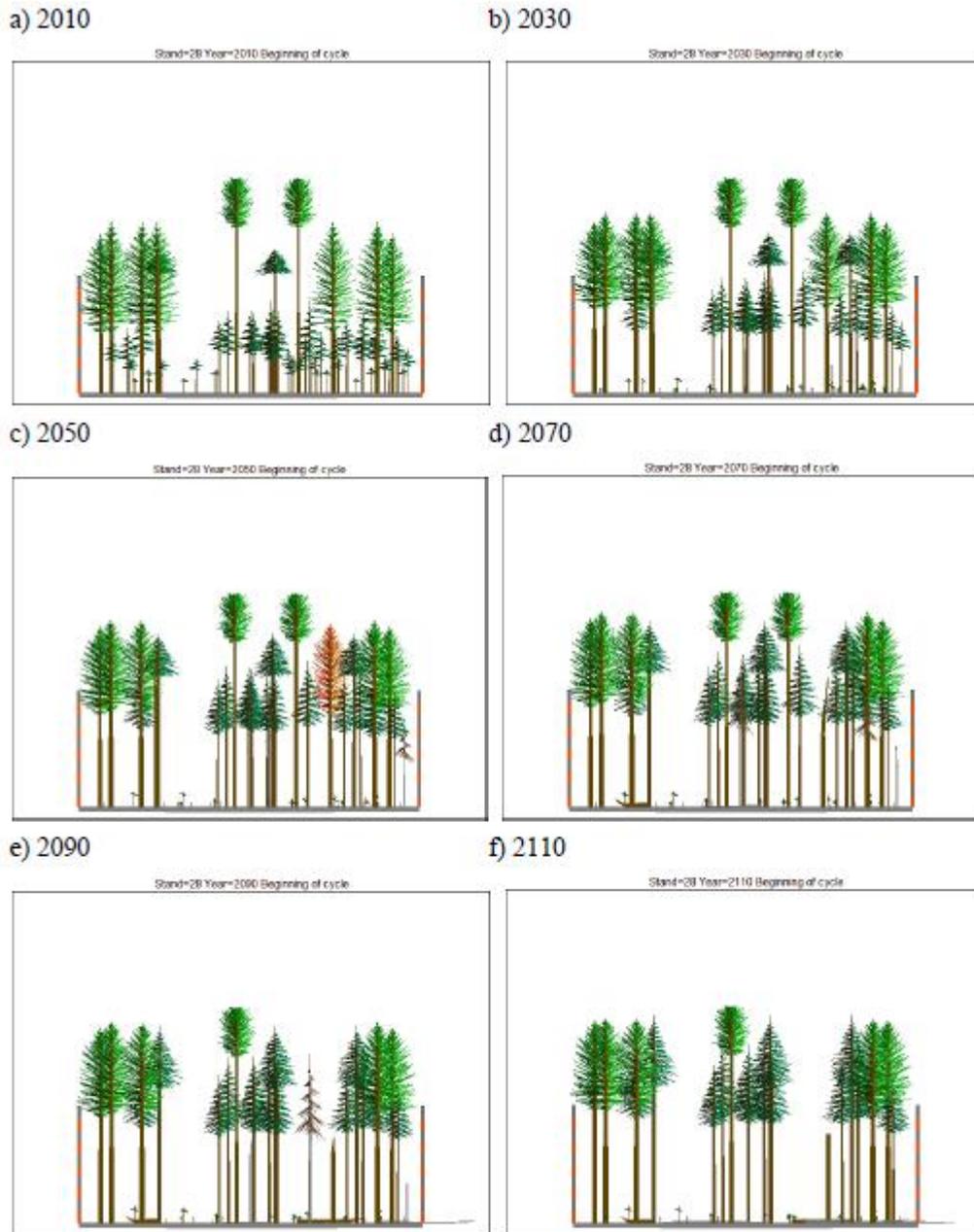


Figure A2-17. SVS profiles of the heavy thinning scenario in the wet high residual 2006 disturbance stand.

Appendix 3 – Hemispherical canopy photographs

a) Plot 303



b) Plot 309



Figure A3-1 a-b. Hemispherical photographs from the wet young conifer vegetation type.

a) Plot 330



b) Plot 350



Figure A3-2 a-b. Hemispherical photographs from the mature wet conifer vegetation type.

a) Plot 316



b) Plot 328



Figure A3-3 a-b. Hemispherical photographs from the old mature wet conifer vegetation type.



Figure A3-4. Hemispherical photographs from the young dry conifer vegetation type (Plot 317).

a) Plot 325 – not a Freda stand



b) Plot 333 – a Freda stand



Figure A3-5 a-b. Hemispherical photographs from the mature dry conifer vegetation type.

a) Plot 321



b) Plot 327



Figure A3-6 a-b. Hemispherical photographs from the old mature dry conifer vegetation type.

a) Plot 322 – This plot is on the edge of the stand. Much of the open area is a parking lot



b) Plot 349



Figure A3-7 a-b. Hemispherical photographs from the mature alder vegetation type.

a) Plot 352



b) Plot 356



Figure A3-8 a-b. Hemispherical photographs from the young mixed deciduous/coniferous vegetation type.

a) Plot 321



b) Plot 355



Figure A3-9 a-b. Hemispherical photographs from the mature mixed deciduous/
coniferous vegetation type.

a) Plot 307



b) Plot 337



Figure A3-10 a-b. Hemispherical photographs from the old mature mixed deciduous/coniferous vegetation type.

a) Plot 202



b) Plot 207



Figure A3-11 a-b. Hemispherical photographs from a) the young CWHxm vegetation type, and b) the old mature CWHxm vegetation type.

Appendix 4 – Tree and understory vegetation species lists

Table A4-1. Common and scientific names of all tree species found in sample plots in 2008 and 2009. Naming convention follows Hitchcock and Cronquist (1976).

Common name	Scientific name
arbutus	<i>Arbutus menziesii</i> Pursh.
bigleaf maple	<i>Acer macrophyllum</i> Prush.
bitter cherry	<i>Prunus emarginata</i> (Dougl.) Walp.
black cottonwood	<i>Populus trichocarpa</i> T. & G.
chestnut	<i>Castanea</i> spp.
Douglas-fir	<i>Pseudotsuga menziesii</i> (Mirbel) Franco.
grand fir	<i>Abies grandis</i> (Dougl.) Forbes
horse-chestnut	<i>Aesculus hippocastanum</i> L.
Norway maple	<i>Acer platanoides</i> L.
Pacific crabapple	<i>Pyrus fusca</i> Raf.
Pacific yew	<i>Taxus brevifolia</i> Nutt.
paper birch	<i>Betula papyrifera</i> Marsh.
red alder	<i>Alnus rubra</i> Bong.
Sitka mountain-ash	<i>Sorbus sitchensis</i> Roemer
Sitka spruce	<i>Picea sitchensis</i> (Bong.) Carr.
western hemlock	<i>Tsuga heterophylla</i> (Raf.) Sarg.
western redcedar	<i>Thuja plicata</i> Donn.
western white pine	<i>Pinus monticola</i> Dougl.
white oak	<i>Quercus alba</i> L.

Table A4-2. Common and scientific names of all understory species found in sample plots in 2008 and 2009. Naming convention follows Hitchcock and Cronquist (1976).

Common name	Scientific name
alaskan blueberry	<i>Vaccinium alaskaense</i> Howell
bedstraw	<i>Galium sp.</i> L.
bleeding heart	<i>Dicentra formosa</i> (Andr.) Walp.
bracken	<i>Pteridium aquilinum</i> (L.) Kuhn.
bullrush	<i>Scirpus americanus</i> Pres.
bunchberry	<i>Coruns canadensis</i> L.
common rush	<i>Juncus effuses</i> L.
creeping buttercup	<i>Ranunculus repens</i> L.
deerfern	<i>Blechnum spicant</i> (L.) Roth.
dull oregon-grape	<i>Berberis nervosa</i> Pursh; also known as <i>Mahonia nervosa</i>
elderberry	<i>Sambucus racemosa</i> L.
English ivy	<i>Hedera helix</i> L.
goat's beard	<i>Aruncus Sylvester</i> Kostel. Sylvan g.; also known as <i>A.diocius</i>
grass	various species growing in maintained lawns
false azalea	<i>Menziesia ferruginea</i> Smith
false lily of the valley	<i>Maianthemum dilatatum</i> (Wood) Nels. & Macbr.
fireweed	<i>Epilobium angustifolium</i> L.
herb-robert	<i>Geranium robertianum</i> L.
Himalayan blackberry	<i>Rubus discolor</i> Weihe & Nees
holly	<i>Ilex aquifolium</i> L.
ladyfern	<i>Athyrium felis-femina</i> (L.) Roth.
nipplewort	<i>Lapsana communis</i> L.
oval leaved blueberry	<i>Vaccinum ovalifolium</i> Smith
thimbleberry	<i>Rubus parviflorus</i> Nutt.
red huckleberry	<i>Vaccinum parvifolium</i> Smith
red osier dogwood	<i>Coruns stolonifera</i> Michx.
salal	<i>Gaultheria shallon</i> Prush
salmonberry	<i>Rubus spectabilis</i> Prush
sedge	<i>Carex sp.</i> L.

Common name	Scientific name
skunk cabbage	<i>Lysichitum americanum</i> Hultén & St. John
spiny woodfern	<i>Dryopteris austriaca</i> (Jacq.) Woynar; also known as <i>D. expansa</i>
stinging nettle	<i>Urtica dioica</i> L.
swordfern	<i>Polystichum munitum</i> (Kaulf.) Presl
three leaved foamflower	<i>Tiarella trifoliata</i> L.
trailing blackberry	<i>Rubus ursinus</i> Cham. & Schlecht.
vine maple	<i>Acer circinatum</i> Pursh
wall lettuce	<i>Lactuca muralis</i> (L.) Fresen.)
water lily	<i>Nymphaea odorata</i> Aiton