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**THE EFFECTS OF URBANIZATION AND HUMAN DISTURBANCE
UPON PLANT COMMUNITY STRUCTURE AND BIRD SPECIES
RICHNESS, DIVERSITY, AND ABUNDANCE IN A NATURAL
FORESTED AREA (FOREST PARK) IN PORTLAND, OREGON**

by

NANCY ELLEN BROSHOT

A dissertation submitted in partial fulfillment of the
requirements for the degree of

DOCTOR OF PHILOSOPHY

in

ENVIRONMENTAL SCIENCES AND RESOURCES: BIOLOGY

Portland State University

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DISSERTATION APPROVAL

The abstract and thesis of Nancy Ellen Broshot for the Doctor of Philosophy in Environmental Sciences and Resources: Biology were presented January 15, 1999 and accepted by the dissertation committee and the doctoral program.

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ABSTRACT

An abstract for the dissertation of Nancy Ellen Broshot for the Doctor of Philosophy in Environmental Sciences and Resources: Biology presented January 15, 1999.

Title: The Effects of Urbanization and Human Disturbance upon Plant Community Structure and Bird Species Richness, Diversity, and Abundance in a Natural Forested Area (Forest Park) in Portland, Oregon.

The effects of urbanization and continual human disturbance on the plant and avian communities of Forest Park and forested lands surrounding Portland, Oregon, were studied. I examined characteristics of plant and avian communities at 25 sites, 24 which were in Forest Park and surrounding areas and one which was in the Ancient Forest Preserve (old-growth stand) northwest of Forest Park. Data were analyzed using multiple regression, ANOVA, and Bonferonni/Dunn. Seven variables were selected representing different urbanization gradients. An additional covariable coded for the old-growth stand, allowing it to be used as a control.

Many tree variables, especially those related to shade-tolerant species, were positively correlated with both the distance from downtown Portland and the number of houses in the surrounding area, and negatively correlated with the distance from the nearest forest edge; however, many shrub and herbaceous variables were negatively correlated with the distance from downtown Portland. Species diversity

for herbaceous and shrub species was greater at more urban sites, but diversity of trees was lower at more urban sites. There were significantly more non-native species of plants in the city section. I found significantly fewer saplings and small trees, especially shade-tolerant species, in the section of Forest Park closest to downtown Portland, although tree mortality was positively correlated with distance from Portland.

Summer bird data revealed significant increases in the abundances of urban and edge species at more urban sites, with concomitant reductions in forest species. There were significantly more ground gleaning birds and short distance migratory species. I also found a significantly greater abundance of birds in the old-growth stand during the winter. This increase was positively correlated with the depth of snow in the nearby Cascade Mountains.

My results indicate that Forest Park is apparently progressing in a normal successional pattern with the exception of the city section. The reduction in shade tolerant saplings and small trees in the city section suggest that rate of succession has been slower at more urban sites. Avian data suggest that urbanization affects bird species abundance and guild composition in the more urban areas.

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of the data, and editing of this document. And finally, I wish to thank my family, especially my husband Morgan and my daughter Christine, for their unwavering support and tolerance throughout this process.

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INTRODUCTION

Forested lands in the Portland Metropolitan area are under ever increasing pressure to be cleared for human use. Concern over loss of the remaining natural areas has prompted several government agencies and conservation groups to buy parcels of forest land to ensure their preservation. One of the largest projects, Metro's Greenspaces Plan, a bond measure, was passed by the public to buy selected "natural" lands for public use, enjoyment, education and preservation. The manner in which these lands are managed will determine if the biological integrity of these natural systems will be maintained. However, there have been few scientific studies addressing such concerns. Indeed, few studies have carefully examined how forested systems change with increasing human pressure.

Portland, Oregon, is fortunate in that since 1948 the city has protected within its boundaries a large forested reserve, Forest Park. In this study, Forest Park was used as a natural laboratory to examine how population dynamics, community structure and biodiversity of Pacific Northwest coniferous forests are changing in relation to human pressure and increased urbanization along an urbanization gradient. This study focused on changes in the plant community and bird species richness, diversity, and abundance as correlated with measures of urbanization.

CHAPTER 1

THE EFFECTS OF HUMANS ON NATURAL ECOSYSTEMS

Throughout the history of ecology, researchers have concentrated their efforts on the investigation of "natural" ecosystems that were unaffected by human activity (Ludwig 1989; McDonnell and Pickett 1990; McDonnell et al. 1993). As people continue to develop land, it is becoming more difficult to find pristine landscapes that have not been influenced by humans. All wilderness areas, even those in remote locations are increasingly affected by human activities (Cole 1995; Cole and Landres 1996; Weaver and Dale 1978). In light of the increasingly dominant role that humans are having upon Earth, it has become crucially important for scientists to investigate the consequences that humans have upon ecosystems.

As human-affected ecosystems become more prevalent, researchers have begun to address the role of humans as components in ecosystems. Most of the work on this subject has been conducted outside of North America, mostly in Europe, where ecosystems significantly unaffected by humans are virtually impossible to find (Becher and Brandes 1985; Blanco and Velásco 1996; Gutte 1990; Hong et al. 1995; Kowarik 1990; Kubantsev and Kolyakin 1995; Natuhara and Imai 1996; Nuorteva 1971; Pysèk 1989; Pysèk and Pysèk 1990; Reijnen and Foppen 1994; Solinska-Górnicka and Symonides 1990; Sukopp 1972; Sukopp 1990; Supuka 1994). However, investigations into the effects of humans on ecosystems have been conducted in this country as well (Airola and Buchholz 1984; Blair 1996; Davis and

Glick 1978; Friesen et al. 1995; Lancaster and Rees 1979; Matlack 1993b; McDonnell et al. 1993; Pouyat and Zipperer 1991; Pouyat et al. 1994; Pouyat et al. 1995a, 1995b; Rowntree 1986; Rudnicki and McDonnell 1989; Sharpe et al. 1986).

One of the problems inherent in studying human-affected ecosystems is how to assess urbanization. For this study, I have defined urbanization as the increasing density of both humans and their accompanying accouterments in cities. Urbanization is accompanied by increasing expanses of impermeable surfaces (e.g., roofs, driveways, roads, and parking lots) as land is cleared and developed. Land cleared of vegetation for development will be greatly altered. But what is the impact of humans upon land that is not cleared of native vegetation -- so-called "green spaces"? Effects of humans on these lands may be direct (e.g., feral cats preying on bird populations) or indirect (e.g., air pollution affecting plant growth rates). In most cases, both direct and indirect factors operate in conjunction with natural factors to produce the community observed at a site. Factors relating to urbanization interact with the ecological community in complex ways. Some factors are easily identified and measured: examples include percent of vegetation removal, the number of buildings, or the distance from structures such as the city, the nearest road, or the nearest building. Other factors are more difficult to ascertain and to quantify. These have been called "subtle factors" (McDonnell et al. 1993) and may play a major role in community structure in urban areas. In addition, the combined effects of many factors may be synergistic and not merely additive (Cole and Landres 1996).

The idea of using a quantifiable gradient such as moisture or elevation to investigate an ecological problem has been a cornerstone in ecological studies for years (Smith

1996; Whittaker 1967; Wilson and Keddy 1986). A similar concept is now being applied in studies on the effects of humans on ecosystems by using various factors of urbanization as a gradient. Examples of quantifiable gradients include the distance from a city (Airola and Buchholz 1984; Pouyat et al. 1994; Pouyat et al. 1995b), the distance from a road (Foppen and Reijnen 1994; Matlack 1993b; Reijnen and Foppen 1994, 1995; Reijnen et al. 1995), and the relative human population density of the surrounding area (Blair 1996; Friesen et al. 1995; Kowarik 1990; Natuhara and Imai 1996; Pouyat et al. 1995b; Pysèk 1989; Pysèk and Pysèk 1990).

Such studies address questions of whether or not urbanization affects an ecosystem and frequently examine how one factor in particular is important. However, urbanization is accompanied by many factors that could impact the ecosystem. West of the Cascade Mountains, in the Pacific Northwest, the natural community that would be expected in most undisturbed areas is a coniferous ecosystem. Forest Park, in Portland, Oregon, provides an excellent area in which to investigate effects of urbanization on a forest adjacent to a city and along an urbanization gradient.

Some effects of urbanization on forests are readily apparent. For example, urbanization is frequently accompanied by increased habitat fragmentation and increases in the amount of "edge" habitat. Human presence is often accompanied by weedy, non-native species of plants and animals which tend to be aggressive competitors, frequently outcompeting native species. Introduced species often have no naturally occurring predators or parasites that could slow their growth and expansion. The result is often rapid overpopulation of exotic species at the expense of native flora and fauna (Bagnall 1979; Cole and Landres 1996; Greller et al. 1990;

Harris 1984; Kowarik 1990; Liebhold et al. 1995; Moran 1984; Sukopp 1990). Examples of exotic species that are problematic in the Portland area include the rock dove (*Columba livia* Gmelin), house sparrow (*Passer domesticus* L.), European starling (*Sturnus vulgaris* L.), English ivy (*Hedera helix* L.), English holly (*Ilex aquifolium* L.), Scotch broom (*Cytisus scoparius* L.), and laurel (*Prunus laurocerasus* L.). As an example, English ivy has become such a problem in forested areas in the Portland area that Portland Parks and Recreation has created a "No Ivy League" to remove ivy and to help educate the public about its impact. In addition to its inevitable negative impact on the native herbaceous plant community, ivy is also believed to have a negative impact on the trees upon which it grows. Fred Nilson (Portland Parks and Recreation, personal communication, March 1994) believes that trees that are covered with ivy are more susceptible to windthrow and that heavy ivy coverage may lead to tree death by covering the plant's apical meristems. As a result, the effects that such exotic species such as English ivy have upon the native plant community may have broader ramifications for the entire community structure of the forest as natural wildlife habitat is altered or lost.

As forested areas outside of Forest Park's jurisdiction are cleared and developed, the once continuous forested landscape of Portland's West Hills is becoming a patchwork of small forest fragments surrounded by cleared and developed (nonforest) landscape. Increased habitat fragmentation causes the abundance and diversity of native species to decrease as forested areas decrease in size (Askins and Philbrick 1987; Blair 1996; Dzwonko and Loster 1988, 1989; Friesen et al. 1995; Gavareski 1976; Harris 1984; Lehmkuhl and Ruggiero 1991; Lynch and Whigham 1984; McIntyre 1995; McLellan et al. 1986; Robinson et al. 1995; Rosenberg and

Raphael 1986; Solinska-Górnicka and Symonides 1990; Terborgh 1989; Weaver and Kellman 1981; Wiens 1994; Wilcove 1985; Zacharias and Brandes 1990; Zipperer 1993).

Fragmentation has been shown to cause significant changes in both the abiotic and biotic conditions in forests due largely to increases in forest edge. Forest edge is subjected to higher diurnal fluctuations of light, wind intensity, and moisture than the forest interior (Chen et al. 1995; Matlack 1993a; Ranney et al. 1981). Plants in the edge habitat are subjected to environmental stresses such as decreased soil moisture, increased light, and increased temperature fluctuations that frequently result in decreased vigor and increased mortality rates (Chen et al. 1992; Chen et al. 1995; Ranney et al. 1981). Edge environments are frequently invaded by exotic species better adapted to the new environmental conditions than are the original native species (Chen et al. 1995; Kroodsma 1984; Moran 1984; Ranney et al. 1981; Rich et al. 1994; Zipperer 1993).

Although this so-called edge effect decreases with distance into the forest, it can extend for a considerable distance into the stand. Chen et al. (1995) demonstrated that the edge effects can extend over 500 meters into an old-growth Douglas-fir forest. Such effects produce a community that is different from the original forest interior community (Rosenberg and Raphael 1986; Temple and Wilcox 1986; Terborgh 1989). Edge effects commonly result in a loss of species that are adapted to the forest interior in favor of "edge species" that enter the area, many of which are common in earlier successional stages (e.g., red alder and song sparrows). The resultant ecotonal community may consequently show greater species diversity as

more common and frequently weedy species of both plants and animals appear, often at the expense of more sensitive forest interior species (Askins 1994; Askins and Philbrick 1987; Donovan et al. 1995a; Friesen et al. 1995; Harris 1984; Lynch and Whigham 1984; McIntyre 1995; McLellan et al. 1986; Ranney et al. 1981; Robinson et al. 1995; Rosenberg and Raphael 1986; Temple 1986; Wilcove 1985). As the degree of fragmentation increases and the size of the resulting forest fragments becomes smaller, the fragments will cease to function as a normal forest ecosystem and will convert entirely to "edge" (Chen et al. 1992; Chen et al. 1995; Harris 1984; Hoover et al. 1995; Matlack 1993b; Matlack 1994; Ranney et al. 1981; Rosenberg and Raphael 1986; Wilcove 1985). The size and shape at which of the forest becomes essentially all edge is called the minimum critical area. Estimates of the minimum critical area for equilateral tracts range from 10 to 20 hectares (Aldrich and Coffin 1980; Lehmkuhl and Ruggiero 1991; Levenson 1981; Lovejoy and Oren 1981; Matlack 1994; McIntyre 1995; Peterken and Game 1984; Ranney et al. 1981) to 100 hectares or more (Brown and Robinson 1996; Chen et al. 1992; Chen et al. 1995; Dobkin and Wilcox 1986; Donovan et al. 1995a; Harris 1984; Hoover et al. 1995; Rosenberg and Raphael 1986; Wilcove 1985). Forest fragments with small forest to edge ratios, regardless of total area, have been reported to function as essentially edge systems (Blake 1983; Dzwonko and Loster 1989; Gotfryd and Hansell 1986; Harris 1984; Johnson et al. 1981; Ranney et al. 1981; Rosenberg and Raphael 1986; Temple 1986; Tilghman 1987; Whitcomb et al. 1981).

A key point is that, for the most part, humans disturb natural ecosystems. Disturbance is most often defined as "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources,

substrate availability, or the physical environment" (White and Pickett 1985). Disturbance, both natural and human-caused, frequently causes communities to revert to earlier successional stages. The degree of setback often depends on both the intensity and frequency of disturbance. Highly disturbed ecosystems will typically be in earlier seral stages than will less disturbed (or undisturbed) ecosystems (Abrams and Nowacki 1992; Dale and Doyle 1987; Hong et al. 1995; Kowarik 1990; Leak and Smith 1996; Matlack 1993b; Petraitis et al. 1989, Swindel et al. 1986).

Variables of urbanization in Forest Park include air pollution, direct trampling or damage to plants, invasion by non-native plant species such as English ivy or English holly, direct disturbance of animals by humans or their pets or by animals that are typically associated with humans such as Norway rats (*Rattus norvegicus*), increased nest parasitism by brown-headed cowbirds (*Molothrus ater* Boddaert), increased predation rates by feral cats (*Felis domesticus*), feral dogs (*Canis domesticus*), or other natural predators such as coyotes (*Canis latrans*) and raccoons (*Procyon lotor*) whose populations react favorably to human presence, and increased traffic noise. Most of these factors are subtle and hard to quantify, but they can affect the structure of the community in important ways.

In addition to spatial gradients, urbanization produces temporal gradients. Many researchers have concentrated on the history of a site in an attempt to determine the reasons for the present community structure (Dale and Doyle 1987; Fain et al. 1994; Glitzenstein et al. 1990; Levenson 1981; McBride and Jacobs 1986; Peart et al. 1992; Peterken and Game 1984; Peterken and Jones 1989; Rudnický and McDonnell 1989; Sharpe et al. 1986; White et al. 1990; Yost et al. 1991; Zipperer et al. 1990).

Disturbance has been positively correlated with the length of human occupation. Europe has had permanent communities and agricultural homesteads for considerably longer than North America and has more disturbed ecosystems (Kowarik 1990; Martin and Clobert 1996; McLellan et al. 1986; Monkkonen and Welsh 1994). Some scientists think that Europe's slower rate of development (almost 5000 years) has allowed plants and animals to adapt to human disturbance (Monkkonen and Welsh 1994), but others question whether we are merely seeing a subset of the original spectrum of species -- those that could adapt to human disturbance (Martin and Clobert 1996; McLellan et al. 1986). Although the historical pattern may never be explained, it illustrates the importance of examining the history of a site, such as Forest Park, in response to the effects of urbanization.

The objectives of this study were to identify a selection of specific, quantifiable factors of urbanization, and to determine the effects those factors have on the plant and avian communities in Forest Park. I investigated the effects of urbanization on plant community structure, and on the breeding and winter resident avian communities.

CHAPTER 2

STUDY AREA

Forest Park is located northwest of downtown Portland, Oregon (Figure 2-1) and covers an area of about 1980 hectares. It is approximately 11.3 km in length and ranges from 0.8 to 2.4 km wide, although the perimeter is very dissected and irregular (Sjulin, J., Portland Parks and Recreation, personal communication, January 1997). The area surrounding the park has various land uses including urban northwest Portland residential neighborhoods at the southern end, rural and suburban lands to the west and north, and urban residential and industrial areas to the east (Portland Parks and Recreation 1995). Forest Park is dissected by several roads. Some, such as Cornell and Germantown Roads, are public roads that have moderate to heavy vehicular use. Others, such as Leif Erikson Drive and Saltzman Road, are presently only open to bicyclists, pedestrians, and vehicles operated by park personnel.

GEOLOGY

Forest Park is located on the east side of the Tualatin Mountains (also called the West Hills) and extends in a northwesterly direction along the Willamette River floodplain. Elevations within the park range from 20 meters near the base of the Tualatin Mountains to 335 meters above sea level at the crest of the Tualatin Mountains near Skyline Boulevard. The Tualatin Mountains are composed of lava flows of the Columbia River Basalt Group that may reach 30 meters or more in thickness (Green 1983). These lava flows are covered by thick deposits of fine textured loess called

Figure 2-1
1994 Ortho Photography

Scale in Miles
0 .9

METRO
800 NE Grand Ave, Portland, OR 97232-2736
Voice 503 787-1742 FAX 503 787-1909
Email drc@metrodel.or.us

Location map

Portland Hills Silt that are believed to have originated as wind deposits during the last two million years with the last major pulse being after the Lake Missoula floods 15,300 to 12,700 years before present (Waite 1985). The deposits range in thickness from a few feet along the lower edge of the Tualatin Mountains to over 30 meters near the crest (Green 1983; Portland Parks and Recreation 1995).

Forest Park has three dominant soil types: Cascade, Goble, and Wauld (Green, 1983). Cascade and Goble soils are often referred to as Portland Hills Silt and share many characteristics. Both soils have silt loam textures and an underlying fragipan (dense silt layer on the soil) that prevents water penetration. The fragipan varies in depth from 51-76 cm deep in Cascade soils to 76-114 cm deep in Goble soils. Wauld soils have a gravel loam texture and no fragipan. Erosion, slumping, and landslide hazards are high with the Cascade and Goble soils and slight to high with the Wauld type soil (Green 1983). Much of the park includes slopes that exceed 30 percent grade. When soil is saturated during wet winter months, severe landslide potential exists in many areas (Bureau of Planning 1992).

Forest Park is dissected by many streams that drain mostly northeast toward the Willamette River. Major watersheds in the park include Balch, Saltzman, Doane, and Miller Creeks (Bureau of Planning 1992).

CLIMATE

Western Oregon is characterized by a maritime climate with mild temperatures, prolonged cloudy periods, relatively cool dry summers, and wet mild winters with moderate to heavy precipitation (Green 1983). In Portland, temperatures range from

an average January temperature of 4.9°C to an average July temperature of 19.8°C. The minimum and maximum temperatures range from -13.3°C in December to 41.1°C in August (Anonymous 1997b). The majority of precipitation falls between October 1 and March 31, mostly as rain. The average annual precipitation in Portland is 1045 millimeters (Anonymous 1997a). Snowfall occurs in the winter months, averaging 130 millimeters per year in Portland and increasing with elevation in the Tualatin Mountains. Severe storms are rare, although one or two ice storms commonly occur annually as the result of cold air flowing west through the Columbia River Gorge (Anonymous 1997a, 1997b; Franklin and Dyrness 1988). Storms with winds over 60 mph are rare in the Portland area, although one occurs every five to ten years. The most damaging wind storm recorded was the October 12, 1962 "Columbus Day Storm." Since that time, five severe wind storms have occurred in the Portland area, with the most recent occurring during the tenure of this study. On December 12, 1995, winds in the Portland area gusted to over 60 mph. Because the strong winds followed a period of higher than normal rainfall, many windthrown trees resulted from water logged soils (Anon 1997c; Taylor 1995). The wind event was followed by one of the wettest winters on record. Winter rainfall levels were 141 percent of normal in Portland. The heavy rains were followed by a cold snap that resulted in large accumulations of snow in the mountains and icy conditions in the Portland area. Then, on February 6, 1996, a subtropical jet stream brought warm air and more record rainfall. The Portland Airport recorded a four day total of 177.8 millimeters of rain -- 50.8 millimeters above the previous record. The combination of weather caused rivers to rise rapidly, resulting in flooding (Taylor 1996). This storm complex resulted in frequent episodes of soil slumpage and landslides in Forest Park (Burns and Fiedorowicz 1997).

PLANT COMMUNITY

Forest Park lies in the western hemlock (*Tsuga heterophylla* Raf. Sarg.) vegetation zone (Franklin and Dyrness 1988). Although western hemlock and western red cedar (*Thuja plicata* Donn.) are considered the climax species for the area, most forests are dominated by Douglas-fir (*Pseudotsuga menziesii* Mirbel Franco) (Franklin and Dyrness 1988). Douglas-fir is capable of living on a site for over 500 years. As a result, old-growth forests often are dominated this species. In addition, Douglas-fir is a fire-adapted species that can survive fires that kill other tree species such as western hemlock and western red cedar (Agee 1991; Franklin and Dyrness 1988; Spies and Franklin 1991). In western Oregon, the return time for fire is estimated to have been 230 years before Europeans began a policy of fire suppression (Agee 1991).

Today, much forested land in western Oregon is covered with early successional (seral) stands that have followed clearing, logging, and/or fires. The composition and density of the seral forests depend on the type of disturbance, the available seed source, whether or not the stand was replanted, and environmental conditions. It is common to find dense stands of essentially even-aged Douglas-firs that result from extensive planting of the species after logging and/or fires. Douglas-fir is a shade-intolerant tree species, meaning that it cannot germinate and grow in the shade of the forest understory; hence, it cannot regenerate in its own presence if the canopy is continuous. Shade-tolerant tree species (e.g., western hemlock) can grow in the shade of larger trees and are more likely to survive in the understory. Western hemlock, the climax species, often exists as an understory tree in these early

successional forests, but it will not begin to dominate until tree mortality begins to open the stand, often 100 to 150 years into stand development (Franklin and Dyrness 1988). In the Portland area, many sites are dominated by shade-intolerant deciduous tree species such as bigleaf maple (*Acer macrophyllum* Pursh) and red alder (*Alnus rubra* Bong) due to the effects of continual disturbance by logging and/or fire (Houle 1996). Douglas-fir may be present, but usually it is not dominant. Shade-tolerant conifers such as western hemlock, western red cedar, and Pacific yew (*Taxus brevifolia* Nuttall) occasionally occur but remain suppressed until mortality begins to open the sites at 50 to 100 years of age (Franklin and Dyrness 1988; Franklin and Pechanec 1967; Houle 1996; Newton et al. 1967).

True climax forests -- forests composed of and dominated by shade-tolerant climax tree species -- are rare in the Pacific Northwest's western hemlock zone due to the extremely long life span of Douglas-fir and the frequent occurrence of fire and other disturbance. Old-growth forests -- forests in later stages of development -- do occur, but most in the Portland area are dominated by Douglas-fir. During the last decade, forest scientists have attempted to define an old-growth forest. An exact definition is elusive and varies with locality (Franklin and Spies 1991a; Franklin and Spies 1991b; Norse 1990). Among the criteria for defining an old-growth forest are the presence of many trees over 100 centimeters in diameter (including shade-tolerant species), a large number of shade-tolerant trees, a large number of large snags (dead standing trees), and a large amount of downed woody debris (Franklin and Spies 1991a; Franklin and Spies 1991b; Norse 1990). Throughout the 1980's, the Forest Service routinely defined areas of only four hectares as sufficient to meet old-growth criteria. Today, researchers have stressed the need for a minimum area of at least 32

hectares to overcome edge effects (Marcot et al. 1991, Norse 1990, Old-Growth Definition Task Group 1986). In the area surrounding Portland, no large expanses of old-growth forests remain, as most were logged shortly after the influx of European settlers. However, a few remnant stands of mature to old-growth forest can be found in areas such as Oxbow Park, the Ancient Forest Preserve, and Forest Park. All of these sites are dominated by large Douglas-firs, but large western hemlocks and western red cedars also are common.

CHAPTER 3

PAST INFLUENCES OF HUMANS ON FOREST PARK

Humans are believed to have inhabited the Pacific Northwest for about 10,000 years. The native people used many of the resources in the forest, although it is believed that they did not modify the coniferous forests substantially (Portland Parks and Recreation 1995). There are some reports of native people setting fires, but it is believed that lightning was the major source of ignition of forest fires in the area (Agee 1991).

Records from the expedition of Lewis and Clark (1805-1806) described the hillsides that are currently the location of Forest Park as being composed of several species of fir that grew to very large heights and diameters. Journal entries also reported that the Portland area was inhabited by about 4,000 native people. The nearest village housed about 100 inhabitants and was located on the bluff across the Willamette River from Forest Park in the area occupied today by the University of Portland (DeVoto 1953).

In the mid-1800's, settlers constructed trails across the Tualatin Mountains to bring crops to the settlements of Linnton (established 1843), Springville (established 1843), and Portland (established 1845) from the Tualatin Valley. These trails included what later became Germantown, Springville, Cornell, and Newberry Roads. In the 1850s, the government deeded much of the Tualatin Mountain land to settlers under the Donation Land Claims Act. The more level land was rapidly

converted to agricultural and residential uses. The first subdivision, Mountain View Park, was recorded in 1882. By 1910, several housing subdivisions had been marketed. Most of these developments avoided steep slopes (Munger 1960).

In 1903, the Municipal Park Commission of Portland hired John Charles Olmsted and Frederick Law Olmsted to conduct a city planning study. The Olmsted brothers proposed a system of parks for the city of Portland that included the recommendation that the eastern slope of the Tualatin Mountains be preserved as a forested park. In 1912, E.H. Bennett, in the Greater Portland Plan, had reiterated the Olmsteds' earlier proposal. Both proposals were ignored by the city for many years (Munger 1960).

Hillside Drive, now called Leif Erikson Drive, was built in 1915. This road connected N.W. Thurman Street and Germantown Road. Several large subdivisions were platted along the drive. However, the steep terrain of the hillside made building and road construction and maintenance difficult. Many land owners refused to pay assessments for property they could not use. Slightly over 1000 hectares of the land was subsequently forfeited to the City of Portland or Multnomah County (Munger 1960).

The original forests were exploited early by the European settlers. The wood was used mostly for fuel for steamboats, mills, and homes, and for building materials. The forests closest to the river were exploited first and to the greatest extent. As the easily accessible forests were depleted, logging expanded upslope. In 1914, the city of Portland ran a wood-cutting camp that provided work for the unemployed and wood for the poor. In 1937, a second wood-cutting camp was operated between

Springville and Saltzman roads west of Leif Erikson Drive (Munger 1960). In addition to the city sanctioned wood-cutting camps, widespread illegal cutting of trees occurred throughout the area (Keil, W., retired forester for Portland Parks and Recreation, personal communication, Dec. 1997). Most of the logging that was done in the park was by clearcutting. Today, many areas of the park still have large rotting stumps with springboard "notches" and skid roads as evidence of logging. Many logging operations were finished with slash burns that often resulted in forest fires (Munger 1960).

In 1944, the Portland City Club appointed a committee to study land use alternatives for the east slope of the Tualatin Mountains. This group, which ultimately became known as the Forest Park Committee of Fifty, strongly supported the creation of a large forested park for the "benefit of the community". In 1947, the committee recommended that the city dedicate over 2300 hectares to park purposes. The land that had been forfeited in the failed Hillside Drive subdivision ultimately became a major part of Forest Park. City and county lands within the proposed boundaries were withdrawn from sale, city land that was held by the Assessment Collection Division was transferred to the Park Bureau, and similar county land was declared park and deeded over to the city. Dedication ceremonies for Forest Park were held on September 25, 1948 (Munger 1960).

The two largest fires in Forest Park occurred in 1940 and 1951. In August 1940, the Bonny Slope fire consumed approximately 405 hectares (1000 acres) on both sides of the Tualatin Mountain ridge. The extent of the burn into Forest Park is uncertain. In August 1951, an even larger fire started near the St. Johns Bridge, ultimately

burning over 480 hectares (1200 acres) (Keil, W., retired forester for Portland Parks and Recreation, personal communication, Dec. 1997; Munger 1960). The 1951 fire was extremely hot and devastated 260 hectares (640 acres) of land in the central portion of the park. It raised fire awareness, however, and the following years brought the construction of the fire lanes, water tanks, and lookout towers. Two areas of the park were salvage logged the year following the blaze (near firelanes 3 and 5). Grade school children helped replant the forest. Quality trees were scarce, however, and anything that could be purchased was planted including ponderosa pine (*Pinus ponderosa* Dougl) and Port Orford cedar (*Chamaecyparis lawsoniana* Parl) (Keil, W., retired forester for Portland Parks and Recreation, personal communication, Dec. 1997).

The U.S. Forest Service produced the first management plan for Forest Park in 1950. This report, "Management Recommendations for Portland City Forest Park", outlined procedures and recommendations for the park and included plans for recreational development such as campgrounds and trails, timber harvest, planting, and forest protection against fire and insect damage. Logging was seen as a means of providing revenue for the park. It was believed that the park was capable of sustained yield harvests. Several timber harvests, some illegal and some legal, took place in 1951 and 1952 (Munger 1960). According to William Keil (retired forester, Portland Parks and Recreation, personal communication, Dec. 1997), the only "legal" logging operation in the most northern section of the park (the section furthest from the city) between 1952 and 1956 occurred near firelane 13. During this same time, no logging occurred in the most southern section of the park (the section closest to the city) and only salvage logging was done in the central portion of the park. The

last logging operation was a small salvage logging project in the Macleay and Holman areas of the park undertaken after the Columbus Day storm in 1962 (Keil, W., retired forester for Portland Parks and Recreation, personal communication, Dec. 1997; Sjulín, J., Portland Parks and Recreation, personal communication, Jan. 1997).

In November of 1976, a new "Management Plan for Forest Park" was adopted by the City of Portland. This plan stressed conservation and preservation of the "naturalness" of the park. Although the goal was an "undisturbed, natural forest", human disturbances were permitted such as the removal of trees to open scenic vistas, the salvage of damaged trees, clearing of rights of way, and removal and sale of forest products (Portland Parks and Recreation 1995).

In February 1995, the City of Portland adopted the "Forest Park Natural Resource Management Plan". This new plan outlined plans and goals for Forest Park's future. The plan was designed to accommodate recreational uses of the park while ensuring protection of the park's forest ecosystem. The goals of the plan include the following: to "establish and maintain a natural and stable ecosystem within the park and surrounding natural areas. It is vital that the flora, fauna and habitat elements remain as free as possible from disturbance related to human activity" (Portland Parks and Recreation 1995).

CHAPTER 4

GENERAL METHODS AND ANALYSIS OF DATA

I selected 24 study sites in Forest Park and adjacent forested areas that are managed with Forest Park by Portland Parks and Recreation. One additional study site was located in a stand of mature (called old-growth for the remainder of this study) Douglas-fir approximately 4 kilometers northwest of Forest Park, a short distance east of McNamee Road (Figure 2-1, page 11). This site was purchased by the Friends of Forest Park in 1993 but is being transferred to METRO, the regional government, for management. This forested area, called the Ancient Forest Preserve, is approximately 12 hectares. It is almost entirely surrounded by a clear cut that was replanted in 1991, although some land with second growth remains adjacent to the stand. Although this stand has some very large Douglas-firs, it is not technically classified as old-growth because there are too few very large trees, too few large snags, too few large downed logs, and, most important, the stand is too small (Franklin and Dyrness 1988; Franklin and Spies 1991b; Spies 1991).

Each study site was randomly located. The procedure used was to stratify the park by dividing it approximately into thirds and randomly assigning eight sites to each third (Figure 4-1). Thereby, the sites were distributed along the perceived urban-rural land use gradient. The most southeasterly third of the park is closest to the city (i.e., the city section). The most northwesterly third is farthest from the city (i.e., the far section). A northeast to southwest urbanization gradient could be a factor in this study because Portland's Northwest industrial area lies adjacent to the northeast

Figure 4-1
Site Locations and
Research Sections

- Parks/Open Space
- Forest
- Shrub/Scrub
- Meadow
- Other

Scale in Miles
0 .9

METRO

800 NE Grand Ave, Portland, OR 97232-2738
Voice 503 787-1742 FAX 503 787-1909
Email drc@metrodot.or.us

Location map

boundary of Forest Park for several kilometers. However, there were approximately the same number of sites on both the northeast (13 sites) and the southwest (11 sites) sides of the park so this alternative could be examined.

Three, 30 meter long transects were established at each study site. At most sites, transects were established in a parallel fashion 20 meters apart. However, sites 7, 18, 36, and 60 were located along ridges. At these four sites, transects were laid end to end along the ridgeline to reduce variance of the sample. At each site, vegetative structure and avian species abundance and richness were measured.

All trees and saplings, living or dead, within a 500 square meter quadrat (8.3 meters on either side of the 30 meter transect) were measured and counted. I defined trees as plants that were greater than 3 meters tall with a dbh (diameter at breast height, 1.4 meters above the ground) greater than or equal to 10 centimeters. I defined saplings as plants that were between 0.5 meters and 3 meters in height or, if 3 meters or taller, had a dbh of less than 10 centimeters, but that were members of arboreal species. Tree diameter was measured with a diameter tape (± 0.1 centimeter). Tree height and the height to the lowest living branch were measured using a rangefinder (± 10 centimeters) for each tree or sapling and used to calculate the live crown ratio by dividing the length of the bole with living branches by the height of the tree. In addition, randomly selected trees and saplings within each quadrat were cored and the core used to estimate tree age and increment growth rate. All tree seedlings that were present on the transect were identified to species and counted.

All shrubs within a 100 square meter quadrat (1.67 meters on each side of the transect) were measured and counted. I defined shrubs as woody species greater than one but rarely more than four meters in height. Some species, such as vine maple (*Acer circinatum* Pursh.) and English holly (*Ilex aquifolium* L.), that have the capacity to exceed the height limitation, were classified as shrubs due to their habit of sending up numerous small shoots and their relatively small size at maturity on my sites. The diameter of the stem at ground level was measured using a diameter tape for all shrubs. For shrubs that sent out numerous shoots, a measurement of the diameter of the entire clump was made. In addition, the height of each shrub was measured.

Herbaceous species were recorded as percent cover along each 30 meter transect. I defined herbaceous species as all non-woody species of any height, or woody species that rarely exceeded one meter in height at my sites (e.g., salal (*Gaultheria shallon* Pursh), Oregon grape (*Berberis nervosa* Pursh)).

Researchers have used various measures to estimate the extent of urbanization. Measures that have been cited in the literature include the distance from a city (Airola and Buchholz 1984; Pouyat et al. 1994; Pouyat et al. 1995b), the distance from a road (Foppen and Reijnen 1994; Matlack 1993b; Reijnen and Foppen 1994; Reijnen et al. 1995), and the relative population density of the area surrounding the study site (Blair 1996; Friesen et al. 1995; Kotz 1990; Kowarik 1990; Pysèk 1989; Pysèk and Pysèk 1990). I thought that these variables were insufficient to fully investigate the impact of human activities on Forest Park. Therefore, additional variables were used that I deemed representative of the degree of forest fragmentation, edge effects,

possible pollution effects, and the degree of human use of the park and surrounding landscape. From each site, I estimated the distance on U.S.G.S. 7.5 minute series topographic maps (Portland Quadrangle, Linnton Quadrangle, and Sauvie Island Quadrangle) from the Multnomah County Courthouse, located at 1021 S.W. Fourth Avenue in Portland, Oregon, to each site as an indication of the distance from downtown Portland. I determined all other distances either from infrared aerial photographs taken in 1989 that were provided by the Portland State University Geography Department or from an orthophoto map (1 inch = 1200 feet) created from 1994 data by METRO. On the 1994 map, I measured the distance from each site to the nearest edge of contiguous (unfragmented) forest, as well as the distance to any forest edge (including small openings) that was visible on the photos or orthophoto map. I also noted the usage category of that edge (e.g., residential, industrial, farm, road, etc.). I assessed the amount of edge habitat by measuring the distance from each site to the nearest public road and noting the degree of usage of that road, to the nearest building and its usage (e.g., industrial or residential), and to the nearest point of the Northwest industrial section of Portland bordering the park. Because Forest Park and the forested lands surrounding it vary in width, I measured the width of contiguous forest along the narrow axis of the park (northeast to southwest) relative to each site. I also measured the distance from each site to both the northeastern and southwestern edges of contiguous forest and used these measurements to calculate the percent of the distance that each site was from the center of the forested area. Because it is recognized that a minimum critical area is important in maintaining species diversity, land use in a 20 hectare and a 100 hectare circular area around each site was determined (e.g., rural, forest, residential, industrial). Most of these measurements were done as percentages, but I counted the number of buildings

within a 100 hectare circular area to determine housing density. I also determined the amount of forest interior habitat for the three sections of Forest Park (i.e., the city, middle, and far sections) assuming that the edge effect can extend either 250 meters or 500 meters into the forest .

Disturbance has been shown to dramatically affect plant communities (Agee 1991; Dale and Doyle 1987; Fain et al. 1994; Glitzenstein et al. 1990; McBride and Jacobs 1986; Peart et al. 1992; Peterken and Game 1984; Rudnický and McDonnell 1989; Sharpe et al. 1986; Solinska-Górnicka and Symonides 1990; White et al. 1990; Yost et al. 1991; Zipperer et al. 1990). I attempted to determine the last known date of logging or fire, and if possible, the size, severity, and frequency of such past disturbance at each site using aerial photographs, old maps, data provided by Portland Parks and Recreation, ages of tree cores, and interviews with individuals who were well acquainted with park history.

I attempted to determine the immediate and direct effects that humans inside the park may be having at each site. The number of humans on foot and on bicycle, and the number of dogs that passed each site within a ten minute period were counted. Each count lasted ten minutes and was done simultaneously with the bird counts. Fourteen human usage counts were conducted at each site over a four year period (1993-1996). I also counted the number of aircraft that passed overhead in the same ten minute period. The background noise level at each site over the same ten minute period was judged subjectively using a scale of zero to ten. A zero was recorded when no human-associated noise was detected in the background. A ten was recorded when the background noise was so loud and continuous that only the

loudest natural sounds could be heard. Human-associated noise included automobiles, trucks, sirens, whistles, voices, barking dogs, and industrial noise, but did not include aircraft as they were recorded separately. An estimate of total human effects was calculated by summing the numbers of human usage (determined by the number of humans and dogs), overhead noise (the number of aircraft), and background noise (the level of background sounds).

In total, I measured 29 different variables related to urbanization. Twenty-five of these were continuous. I used multiple regression to test for relationships between urbanization variables (i.e., independent variables) and measurements of vegetative or avian community structure (i.e., dependent variables). Multiple regression tests for the importance of several different factors simultaneously (Abacus Concepts 1992; Draper and Smith 1966; Johnson and Wichern 1982; Jongman et al. 1995; Sokal and Rohlf 1969; Steel and Torrie 1960). It is unlikely that a single variable of urbanization is uniquely important. In most cases, different variables of urbanization jointly affect one another to produce the observed results (Draper and Smith 1966; Johnson and Wichern 1982; Pimentel 1979). Before multiple regression analysis could be used, it was necessary to determine which of the urbanization variables were both important and independent of one another (Abacus Concepts 1992; Draper and Smith 1966; Johnson and Wichern 1982; Jongman et al. 1995; Morrison 1976; Pimentel 1979; Steel and Torrie 1960). Using a correlation matrix, I reduced the number of useful variables of urbanization to seven by selecting variables that had less than 60 percent collinearity. I also selected variables that I felt best measured different components of urbanization. Of the seven variables selected to be used in this study, the distance from the nearest forest edge and the percent of surrounding

forest were the two most highly correlated with a correlation coefficient of 0.57 (see Table 4-1 for correlation matrix). This level of collinearity is acceptable in the regression model used (Abacus Concepts 1992). The seven urbanization variables that were chosen functioned as the independent variables in all multiple regression analyses used throughout the study.

The seven independent variables of urbanization used in all the multiple regression analyses (and their abbreviations used throughout the paper) were: the distance from the Multnomah County Courthouse (PDX), the distance from the nearest road (ROAD), the distance from the nearest section of the Northwest industrial area (NWI), the distance from the nearest forest edge of any type that was large enough to be observed on the aerial photographs (approximately 3 m) (EDGE), the number of houses in the surrounding 100 hectares (HOUSE), the percentage of forest in the surrounding 100 hectares (FOREST), and the proportionate distance from the center of contiguous forest to the forest edge (CENTER). Each variable represents some degree of influence by humans and helps define a gradient of urbanization. A higher degree of human influence at a site could be expected to be positively correlated with the proximity to Portland, the nearest road, the Northwest industrial section, the nearest forest edge, and the number of surrounding houses; and inversely correlated with the percent of surrounding forest and the distance from the center of the forest.

To meet the assumptions of normality, some of the independent variables were transformed so that they exhibited a normal distribution when graphed (Table 4-2) (Draper and Smith 1966; Jongman et al. 1995; Sokal and Rohlf 1969; Steel and Torrie 1960; Tabachnick and Fidell 1996). These transformations were used

Table 4-1. Correlation matrix for the seven urbanization variables. PDX is the distance from downtown Portland; ROAD is the distance from the nearest road; NWI is the distance from the nearest section of the Northwest industrial area; EDGE is the distance from the nearest forest edge of any kind that is evident on aerial photos; FOREST is the percent of forest in the 100 hectare circular area surrounding the site; CENTER is the percent of the distance from the center of the park to the edge of contiguous forest; and HOUSE is the number of houses in the 100 hectare circular area surrounding the site.

	ROAD	NWI	EDGE	FOR100	CENTER	HOUSE
PDX	0.385	0.473	0.108	-0.172	-0.219	-0.427
ROAD	—	-0.028	0.345	0.524	-0.428	-0.498
NWI	—	—	-0.111	-0.439	-0.346	0.008
EDGE	—	—	—	0.566	-0.111	-0.234
FOR100	—	—	—	—	-0.385	-0.532
CENTER	—	—	—	—	—	0.105

Table 4-2. The seven independent variables, their abbreviations as utilized throughout the paper, and the transformations done to meet assumptions of normality. These transformations were used in all statistical analyses.

<u>Urbanization Variable (Abbreviation)</u>	<u>Transformation</u>
Distance from downtown Portland (PDX)	none
Distance from the nearest road (ROAD)	none
Distance from the Northwest industrial area (NWI)	square root (NWI)
Distance from the nearest forest edge (EDGE)	square root (EDGE)
Percent of forest in the surrounding 100ha (FOREST)	arcsin (FOREST)
Percent of distance from center of forest (CENTER)	square root (CENTER)
Number of houses in surrounding 100ha (HOUSE)	ln (HOUSE + 1)

consistently in all data analyses. Dependent variables that were not normally distributed were also transformed (Draper and Smith 1966; Jongman et al. 1995; Sokal and Rohlf 1969; Steel and Torrie 1960; Tabachnick and Fidell 1996). The various transformations are reported in the results as they are presented throughout the paper. Species or categories that were detected too infrequently or were absent from too many sites were considered rare and were omitted from statistical analyses. The exact method of determining rare species varied between different types of data and will be reported for each.

In addition, an eighth independent variable was added. This was a covariable that was highly correlated (correlation coefficient of 0.83) with the old-growth stand. This variable coded all sites as being either in the old-growth stand or in Forest Park and is abbreviated as OG throughout the paper. The addition of this covariable to all multiple regression analyses allowed me to determine whether or not significant findings were due to the urbanization variables or were the result of the old-growth stand being very different from the other sites. The addition of this variable also allowed the old-growth stand to be used as a control against which Forest Park could be compared (Perrin, N., Professor of Psychology, Portland State University, personal communication, Nov. 1998).

The results of each significant regression are presented in tabular form by the degree of significance, whether the correlation was negative or positive, the squared multiple regression coefficient (R^2), and the p-value for the regression. The R^2 value represents the proportion of the dependent variable's variability that is explained by the independent variables. The p-value is the probability that R^2 is significantly

greater than zero as determined by t-test. All statistically significant ($p < 0.05$) dependent variables that had R^2 values greater than or equal to 0.25 were considered important and are presented in the tables. All statistical tests were conducted using Statview, version 4.5 (Abacus Concepts 1992).

All dependent variables that were significant by multiple regression were analyzed further by analysis of variance (ANOVA) using section of the park as a nominal variable (Steel and Torrie 1960). This variable divided the park into three sections relative to the distance from downtown Portland (city end, middle section, and far end) with a fourth category for the old-growth stand. These divisions were in accord with the manner in which I initially selected sites (figure 4-1). Bonferroni/Dunn values were used to determine significance among the categorical variables in each ANOVA. This particular test was used due to its strict method of discriminating among variables (i.e., it is a very conservative test) and because I did not have equal numbers of samples in each group (Abacus Concepts 1992; Johnson and Wichern 1982; Morrison 1976). All determinations of significance are reported at the 95 percent confidence interval.

CHAPTER 5

THE EFFECTS OF HUMAN DISTURBANCE AND URBANIZATION UPON THE STRUCTURE OF THE PLANT COMMUNITY

Many scientists have examined how land use history affects plant community structure in forests (Abrams and Nowacki 1992; Agee 1991; Dale and Doyle 1987; Fain et al., 1994; Fralish et al., 1991; Glitzenstein et al. 1990; Greller et al. 1990; Gutte 1990; Kowarik 1990; Levenson 1981; McBride and Jacobs 1986; Nowacki and Abrams 1992; Palik and Pregitzer 1992; Peart et al. 1992; Peterken and Game 1984; Peterken and Jones 1989; Rudnický and McDonnell 1989; Sharp et al. 1986; Sukopp 1972; Weaver and Kellman 1981; White et al. 1990; Whitney 1986; Yost et al. 1991; Zipperer et al. 1990). Some investigators found that forests resulting from clearcuts or burns appear to be very similar to pre-disturbance forests (Glitzenstein et al. 1990; Leak and Smith 1996; Weaver and Kellman 1981; White et al. 1990), whereas others have reported that present day regrown mature forests represent very different communities from pre-settlement forests (Abrams and Nowacki 1992; Dzwonko and Loster 1989; Fralish et al. 1991; Greller et al. 1990; Nowacki and Abrams 1992; Palik and Pregitzer 1992; Peterken and Game 1984; Rudnický and McDonnell 1989; Sachse et al. 1990; Seidling 1990; Yost et al. 1991). Both fire management (Abrams and Nowacki 1992; Fralish et al. 1991; Nowacki and Abrams 1992; Palik and Pregitzer 1992) and the introduction of exotic species (Greller et al. 1990; Sachse et al. 1990; Seidling 1990; Yost et al. 1991) have been implicated as reasons for differences in forest structure. Other causes proposed include the effects

of trampling, predation, and pollution (Kubikova 1990; Rudnický and McDonnell 1989).

As urbanization increases, frequently accompanied by increased fragmentation of the existing forest, the number of native species decreases, whereas the number of exotic species increases (Bagnall 1979; Becher and Brandes 1985; Greller et al. 1990; Johnson et al. 1981; Kowarik 1990; Levenson 1981; Moran 1984; Seidling 1990; Sukopp 1990; Yost et al. 1991; Zacharias and Brandes 1990; Zipperer et al. 1990). Ecosystems that have been subjected to frequent and/or severe disturbances have been reported to exhibit reduced species diversity (Airola and Buchholz 1984; Davis and Glick 1978; Dzwonko and Loster 1988; Hoehne 1981; Rao et al. 1990; Sharp et al. 1986; Solinska-Gornicka and Symonides 1990; Swindel et al. 1986). However, moderate levels of human disturbance may cause reductions (Airola and Buchholz 1984; Bagnall 1979; Davis and Glick 1978; Dzwonko and Loster 1988; Fugate 1994; Greller et al. 1990; Iida and Nakashizuka 1995; Kotz 1990; Pysěk and Pysěk 1990; Rao et al. 1990; Seidling 1990; Solinska-Górnicka and Symonides 1990) or increases in the species diversity of vascular plants (Dzwonko and Loster 1988, 1989; McBride and Jacobs 1986; Moran 1984). These seemingly contradictory reports may be explained by the "intermediate disturbance hypothesis". According to this hypothesis, moderate levels of disturbance frequently lead to increased species diversity due to increased heterogeneity of disturbed habitat versus nondisturbed habitat (Connell 1978; Huston 1979; Petraitis et al. 1989). However, as disturbance increases and the habitat becomes more homogenous, diversity begins to decrease again. This hypothesis has been supported by a number of investigators who have

reported that plant species diversity initially increased and then decreased as disturbance intensified (Armesto and Pickett 1985; Hoehne 1981; Kowarik 1990).

In my research, causes of changes in species diversity may be complicated because Forest Park may function ecologically as a peninsula rather than an island. Many workers have observed that species diversity decreases toward the tip of a peninsula (Gilpin 1981; Milne and Forman 1986; Simpson 1964). However, in some studies, diversity of certain reptilian or mammalian groups did not seem to fit this model and were better explained by environmental gradients (Busack and Hedges 1984; Lawlor 1983; Seib 1980). Milne and Forman (1986) determined that plants along peninsulas in Maine supported the general hypothesis, but they determined that this finding was more closely related to environmental heterogeneity than to any peninsular effect.

Not only does the number of species change, but scientists investigating the effects of urbanization on plant community structure have also reported wide varieties of changes in stand characteristics. The relative importance values (i.e., the relative contributions of species to the entire community calculated by summing the relative density, the relative frequency, and/or the relative coverage or basal area of trees (Brower et al. 1990)) of tree species have changed in the New York City Botanic Garden Forest over the past 48 years. *Acer*, *Betula*, and *Prunus* have gained strong footholds in the forest at the expense of two previously dominant trees -- *Quercus* and *Tsuga*. (Rudnický and McDonnell 1989). Similar changes have also been observed in other urban woodland areas (Airola and Buchholz 1982; Yost et al. 1991; Zipperer and Zipperer 1992). Airola and Buchholz (1982) reported that tree

species associated with human settlement have become more important than "normal" forest tree species as disturbance has increased.

The density of trees has been observed to decrease in forests in many urban areas (Airola and Buchholz 1982; McBride and Jacobs 1986; Rudnický and McDonnell 1989), although the opposite has also been reported (Zipperer and Zipperer 1992). Rudnický and McDonnell (1989) found that stem density was higher in the New York City Botanic Garden Forest than it had been 50 years earlier, although it was lower than that found in similar rural forests.

Scientists have also reported that the basal area and/or diameter at breast height (dbh) of trees is reduced in urban or disturbed forests (Airola and Buchholz 1984; James et al. 1979; and Rudnický and McDonnell 1989). Rudnický and McDonnell (1989) found that mortality of dominant tree species exceeded replacement in New York City. The overall effect of disturbance has been for forests to become denser, with smaller diameter trees representing earlier stages in succession. Causes that have been proposed by the researchers for the observed changes include trampling, fires, raking, predation, disease, and pollution (Airola and Buchholz 1984; James et al. 1979; Rudnický and McDonnell 1989).

Besides altering characteristics of the tree community, disturbance also affects other aspects of the plant community. For example, the number of shrub species (richness) decreased (Dzwonko and Loster 1988; James et al. 1979; Levenson 1981; Rao et al. 1990), although shrub cover increased (Dzwonko and Loster 1988; Matlack 1994; Seidling 1990). Dzwonko and Loster (1988) found that altered

distribution of shrubs was not related to the size of the woodland, but rather was correlated with increased anthropogenic pressures. Matlack (1994) reported that the increased density of shrubs in edge environments was caused by enhanced growth of previously existing shoots rather than an influx of new plants. James et al. (1979) observed that certain indigenous species of shrubs were very susceptible to anthropogenic pressures and that native shrubs decreased in frequency, often with a concomitant increase in exotic species. The increase in exotic species has been observed repeatedly (Beissinger and Osborne 1984; Cole and Landres 1996; Dzwonko and Loster 1988; Levenson 1981; Mills et al. 1989; 1991; Yost et al. 1991).

Disturbance has been reported to affect the herbaceous plant community as well. Native herbaceous species richness was higher in undisturbed areas than in disturbed areas (Bagnall 1979; Cole 1995; Cole and Landres 1996; Dzwonko and Loster 1988, 1990; Greller et al. 1990; Hoehne 1981; Kowarik 1990; Kubikova 1990; Seidling 1990; Yost et al. 1991). Areas subjected to large amounts of disturbance exhibited a loss of herbaceous cover (James et al. 1979), but increased disturbance also altered species composition of herbaceous vegetation, frequently resulting in a transition to more acidophilous plants (Greller et al. 1990) and/or to more non-native species (Bagnall 1979; Cole and Landres 1996; Greller et al. 1990; Hoehne 1981; James et al. 1979; Kowarik 1990; Mills et al. 1989, 1991; Seidling 1990; Yost et al. 1991). James et al. (1979) reported that herbaceous, dicotyledonous species of plants appeared to be more sensitive to anthropogenic disturbances than did woody species.

This overview reveals that plant communities change along disturbance gradients as well as urbanization gradients. In many cases, urbanization is believed to cause, directly or indirectly, the observed effects, but this hypothesis remains largely unproved. In this study, I attempt to elucidate changes in a plant community in the Pacific Northwest and correlate those changes with increased urbanization. The study was not designed to investigate specific causes of change, only to determine what changes had occurred.

METHODS FOR MEASURING THE PLANT COMMUNITY

The density and total diameter of all tree, sapling, and shrub species were calculated based on measurements made at each transect. These data were used to calculate the importance value for each tree and shrub species. Abundance data were used to determine the species diversity index using the Shannon-Wiener formula ($H' = -\sum p_i \log_e p_i$, where p_i is the proportion of the "ith" species to all species present) (Brower et al. 1990). The percent of coniferous versus deciduous trees was also calculated for each quadrat, as was the total number of introduced tree species (e.g., mountain ash (*Sorbus aucuparia* L.), hawthorn (*Crataegus monogyna* Jacq.)), and Port Orford cedar (*Chamaecyparis lawsoniana* Parl)) and the number of non-native shrub species (e.g., English holly, Himalayan blackberry (*Rubus discolor* Weihe and Nees), and laurel) as determined using Hitchcock and Cronquist (1974). I also calculated the abundance, density, diameter, and diversity for both shade-tolerant tree species (e.g., western hemlock, western red cedar, and Pacific yew), and shade-intolerant tree species (e.g., bigleaf maple, red alder, and Douglas-fir) (Franklin and Dymess 1988).

The percent cover for all tree and shrub species was measured along each transect by recording the presence and composition of overhead vegetation as the transect was traversed. Areas with no overhead vegetation were classified as sky.

The foliage height diversity (FHD) was determined at each transect in a manner similar to the procedure of MacArthur and MacArthur. The species present at each of eight different heights (ground level, 0.5, 2, 3, 5, 10, 20, and 50 meters) at five locations along each transect were recorded. The foliage height diversity was calculated as a Shannon-Wiener type of diversity index, H' (MacArthur and MacArthur 1961).

The abundance of herbaceous plants along the transect was determined using the percent cover for each species on the transect. Plants covering less than five centimeters in any meter segment were recorded but not included in statistical analyses. Likewise, plants present at the site, but not on the transect or in the quadrat were recorded but not included in statistical analyses. Total cover and the Shannon-Wiener diversity index were calculated for herbaceous plants using the percent cover measurements (Brower et al. 1990). The number of species and the percent cover of non-native herbaceous plant species such as English ivy (*Hedera helix* L.), herb Robert (*Geranium robertianum* L.), lemon balm (*Melissa officinalis* L.), nipplewort (*Lapsana communis* L.), reed canarygrass (*Phalaris arundinacea* L.), vetch (*Vicia hirsuta* L.), wall lettuce (*Lactuca muralis* L.), and common groundsel (*Senecio vulgaris* L.) were determined for each transect (Hitchcock and Cronquist 1974). The number and percent cover of herbaceous species associated with old-growth and mature forests (e.g., vanilla leaf (*Achlys triphylla* Smith D.C.), *Anemone deltoidea*

Hook, sandwort (*Arenaria macrophylla* Hook), coral root orchid (*Corallorhiza* sp. Chat), and foam flower (*Tiarella trifoliata* L) (Spies 1991)) were also used as dependent variables in statistical analyses.

RESULTS

Data Analysis

All dependent variables were analyzed using multiple regression and ANOVA as described in Chapter 4. I examined plant community variables relating to tree species density, richness, and diversity; tree diameter; tree importance; shrub species density, richness, and diversity; shrub height and basal area; herbaceous plant cover, richness, and diversity; and foliage height diversity (FHD). The results are reported in the following sections. Scientific and common names of all tree species present at sites in this study are reported in Appendix A; scientific and common names of all shrubs are reported in Appendix B; and scientific and common names of all herbaceous species are reported in Appendix C. Regression coefficients for plant variables are reported in Appendix D.

Human Usage Gradients and Variables

Each of the seven urbanization variables was examined using ANOVA and Bonferroni/Dunn to determine if there were significant differences among the different sections of the park. Each section was significantly different in terms of distance from downtown Portland (PDX) (Table 5-1). The distance from the nearest road (ROAD) was significantly lower for sites in the city section and the percent of surrounding forest (FOREST) was significantly lower in the old-growth stand. The

Table 5-1. Mean values (standard deviation) of each of the urbanization variables per site for the different sections.

Urbanization variable	City	Mid	Far	OG
PDX	2.8 (0.6) ^a	5.6 (0.7) ^b	8.2 (0.8) ^c	12.6 (-) ^d
ROAD	0.2 (0.1) ^a	0.5 (0.1) ^b	0.4 (0.2) ^b	0.3 (-) ^{ab}
NWI	0.6 (0.2) ^a	0.6 (0.2) ^a	0.6 (0.2) ^a	2.9 (-) ^b
EDGE	0.11 (0.07)	0.13 (0.09)	0.18 (0.12)	0.04 (-)
FOREST	0.8 (0.1) ^a	0.9 (0.1) ^b	0.9 (0.2) ^b	0.5 (-) ^c
HOUSE	51.3 (63.8) ^a	0.8 (0.8) ^b	12.5 (19.5) ^b	3.0 (-) ^b
CENTER	0.4 (0.2)	0.3 (0.1)	0.3 (0.2))	0.1 (-)
n	24	24	24	3

Results of Bonnféroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing (see Table 4-2) .

distance from the Northwest industrial area was greatest for the old growth. There were significantly more houses in the city section of the park, however, there were no significant differences among sections for either distance from the nearest forest edge (EDGE) or distance from the center of the forest (CENTER).

I also analyzed the human usage variables using ANOVA and Bonferroni/Dunn. There was significantly higher human usage at sites in the city section as determined by the number of hikers and joggers, the number of dogs, and total human usage (Table 5-2).

Multiple Regression Analysis of Plant Community Variables

Several variables pertaining to trees correlated significantly with some of the urbanization variables as determined by multiple regression. Many tree variables, especially those related to coniferous or shade-tolerant species, as well as tree species diversity, correlated positively with the distance from downtown Portland (Table 5-3). Most of these same variables were also positively correlated with the number of surrounding houses and were negatively correlated with the distance from the nearest forest edge.

The total diameters (total dbh in cm), the average diameters, and the importance values for some tree species were found to be significantly correlated with urbanization variables by multiple regression (Tables 5-4 and 5-5). Again, many variables that measured coniferous and/or shade-tolerant trees correlated positively with the distance from downtown Portland. The average diameters of both Douglas-fir and cascara were correlated with several urbanization variables. Likewise, the

Table 5-2. Human usage in the different study sections as measured by the mean number of hikers and joggers, the mean number of bicycles, the mean number of dogs, total human site usage[†], the mean number of aircraft, and the mean background noise level per site (standard deviation) for the summer months.

Human Usage Variable	City	Mid	Far	OG
# hikers, joggers ^x	2.1 (4.0) ^a	0.4 (1.5) ^b	0.1 (.5) ^b	0.0 (-) ^b
# bicycles	0.2 (0.8)	0.0 (0.2)	0.0 (0.2)	0.0 (-)
# dogs	0.7 (1.3) ^a	0.1 (0.3) ^b	0.1 (0.3) ^b	0.0 (-) ^b
# aircraft	6.5 (2.5) ^a	6.3 (2.6) ^{ab}	6.8 (2.9) ^a	4.5 (1.7) ^b
Total human site usage ^{x†}	2.8 (4.9) ^a	0.5 (1.5) ^b	0.2 (0.7) ^b	0.0 (-) ^b
Background noise	6.2 (2.1) ^a	4.7 (2.2) ^b	6.1 (1.9) ^a	0.3 (0.5) ^c
n	80	80	80	12

Results of Bonnferoni/Dunn testing is shown for each dependent variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Variables were transformed in the same manner as in the multiple regression analyses to fulfill requirements of normality. A superscript of "x" denotes that the square root of the variable was used in statistical analysis.

[†] Total human site usage = the sum of the total number of people (joggers, hikers, and bicyclists) plus the number of dogs that passed by the site during the 10 minute data collection periods.

Table 5-3. Results of multiple regression analysis for tree density, tree species composition, tree species richness, and tree species diversity. Results are reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported, as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix D.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 †
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	
sqrt(# trees)	+++			--	+			0.34***
sqrt(# conifers)	+++				++			0.38***
ln(# shade tolerant +1)	+++			-				0.35***
sqrt(# cedar)	+++	-		---	++	---		0.56***
# deciduous trees				--				0.28**
sqrt(# alder)				--				0.35***
# tree species	+++			---	+++	---		0.42***
species diversity (H')	+++			-	+++	-		0.38***

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$
+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$
++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$
+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

Table 5-4. Results of multiple regression analysis for total tree diameter (dbh in cm). Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of the multiple regression (R^2) for each dependent variable is reported, as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix D.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 †
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	
sqrt(dbh alder)				--				0.27**
sqrt(dbh cascara)				-			+++	0.38***
sqrt(dbh cedar)	+++			-	++	---		0.56***
sqrt(dbh all trees)	+							0.28**
sqrt(dbh conifers)	+				+			0.36***
sqrt(dbh shade tolerant)	+++							0.33***

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

Table 5-5. Results of multiple regression analysis of the mean diameter (dbh in cm) per tree and importance values (IV) for trees. Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported, as is the significance level (p). n is reported for each category. For regression coefficients, see Appendix D.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 [†]	n
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER		
sqrt (dbh Douglas fir)	--		+++	+++			+	0.36***	250
sqrt (dbh cascara)				--				0.40***	88
sqrt (IV maple)					---		--	0.38***	75
sqrt (IV cedar)	+++			--	++	---		0.55***	75
sqrt (IV conifers)	+				++			0.35***	75
sqrt (IV shade tolerant)	+++							0.31***	75
sqrt (IV deciduous)					--		+-	0.48***	75

[†] Significance level of the squared coefficient of multiple regression; significance levels: *=p<0.05; **=p<0.01; ***=p<0.001
+ or - denotes that the dependent variable is significantly correlated with the independent variable with p<0.05
++ or -- denotes that the dependent variable is significantly correlated with the independent variable with p<0.01
+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with p<0.001

importance values of some coniferous species, as well as shade-tolerant trees, correlated positively with the distance from downtown Portland and the number of surrounding houses. The importance value of deciduous trees was negatively correlated with the old-growth stand.

While many tree variables were positively correlated with the distance from downtown Portland, many shrub variables correlated negatively with this urbanization variable (Table 5-6). Significant variables that were negatively correlated with many urbanization variables included shrub density, shrub species diversity, and many variables related to thimbleberry and red elderberry. Vine maple was the only shrub species to be positively correlated with many of the urbanization variables. Many herbaceous variables were also negatively correlated with the distance from downtown Portland (Table 5-7). Many of these same variables were positively correlated with the distance from the nearest road. Only the amount of bare ground (i.e., lack of herbaceous plant cover) showed opposite correlations.

To calculate foliage height diversity (FHD), I measured percent cover by various species at different heights. The percent cover of sky and the percent cover by cedar, as well as foliage height diversity at several heights were found to be significantly correlated to urbanization variables (Table 5-8). The distance from the nearest road and the distance from the nearest forest edge seemed to be the most important urbanization variables for cover and foliage diversity measurements. Total foliage height diversity was negatively correlated with the distance from the nearest forest edge.

Table 5-6. Results of multiple regression analysis for the density, species richness and diversity, total basal area and height, and importance values (IV) for shrubs. Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R²) for each dependent variable is reported, as is the significance level (p). For all variables, n=75. For regression coefficients, see Appendix D.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	OG
ln (# shrubs +1)	---							0.39***
sqrt (H' shrubs)	--	+						0.25*
sqrt (# non-native shrubs)	-				++			0.27**
sqrt (# red elderberry)	--		+					0.35***
sqrt (# thimbleberry)	---		-		---	+++		0.45***
sqrt (# vine maple)	++		+	---	+	--	+	0.41***
sqrt (total ht shrubs)	--							0.33***
sqrt (IV thimbleberry)	---	+			---	+++		0.53***
sqrt (IV vine maple)	+++		+	---	++	---	++	0.43***

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$
+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$
++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$
+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

Table 5-7. Results of multiple regression analysis for herbaceous plant species cover, and herbaceous species richness and diversity. Results are reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported, as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix D.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 †
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	
sqrt (bracken fern)	-		-	--			-	0.36***
sqrt (fairy bells)	---			+++	++		++	0.54***
sqrt (star-flowered S seal)	--							0.35***
sqrt (sword fern)						++		0.31**
sqrt (inside-out)	---	+++		++				0.29**
sqrt (violet)	---	+++			+			0.49***
sqrt (wild blackberry)	--	+						0.34***
sqrt (% bare ground)	+++	--						0.38***
ln (# species + 1)	---	+						0.37***
sqrt (H')	---	+				-		0.37***
sqrt (# native species)	---							0.30**
sqrt (H' native only)	---	+				-	-	0.35***

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$
+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$
++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$
+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

Table 5-8. Results of multiple regression analysis for percent vegetation cover and foliage diversity (FD) at different heights in the canopy, and foliage height diversity (FHD). Results reported as significant positive (+) or negative (-) correlations with each of the seven independent variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported, as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix D.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R ² †
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	
AT GROUND LEVEL:								
Foliage diversity (FD)	---	+++				-		0.26**
AT 3 METERS:								
sqrt (% cover sky)		+		++		-		0.25**
Foliage diversity (FD)		-		---		++		0.25*
AT 5 METERS:								
% cover cedar	+						+	0.45***
Foliage diversity (FD)				-				0.36***
AT 10 METERS:								
% cover cedar							+++	0.55***
Foliage diversity (FD)			+					0.31***
Total FIID				-				0.25**

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

Plant Community Structure -- Differences Between Sections

Tree density, richness, and species diversity were greatest at sites in the far section of the park (Table 5-9). The far section and old-growth had significantly more shade-tolerant species of trees and more conifers, including western red cedar. There were, however, more deciduous trees in Forest Park than in the old-growth stand, including significantly more red alder in the middle section of Forest Park.

The total diameter of trees was significantly greater at sites in the far section and the old-growth stand than at sites in the city or middle sections of the park (Table 5-10). Douglas-fir comprised a significantly greater amount of the total tree diameter in the old-growth stand. On the other hand, the total diameter of bigleaf maple was greater at all sites in Forest Park, and the total diameter of red alder was significantly greater in the middle section of the park. The average diameter of all trees significantly smaller in the far section (Table 5-11). In addition, there were significant differences in the average diameters for some tree species. Bigleaf maple, red alders, and western red cedar had significantly smaller diameters in the middle section of the park. Douglas-fir was significantly larger in the old-growth stand. Western red cedar was significantly larger in far section of the park and in the old-growth stand.

The importance value incorporates both tree density and tree diameter and varied among the different sections for some tree species (Table 5-12). The importance value of shade-tolerant trees was significantly higher in the far section of the park and the old-growth, as was the importance value of western red cedar. The importance

Table 5-9. Mean tree density, tree species richness, and tree species diversity per 500 m² (standard deviation) for the different sections of the study.

Species	City	Mid	Far	OG
<i>Abies grandis</i>	2.4 (6.9)	0.7 (1.6)	1.1 (2.3)	0.0 (-)
<i>Acer macrophyllum</i>	10.0 (5.4)	11.6 (5.7)	11.2 (5.5)	3.0 (2.0)
<i>Alnus rubra</i> ^x	2.3 (3.3) ^a	5.3 (3.0) ^b	3.3 (3.1) ^a	0.0 (-) ^c
<i>Amelanchier alnifolia</i>	0.1 (0.3)	0.3 (1.0)	0.0 (0.2)	0.0 (-)
<i>Chamaecyparis lawsoniana</i>	0.0 (-)	0.6 (2.9)	0.0 (-)	0.0 (-)
<i>Cornus nuttallii</i>	0.4 (0.8)	0.3 (1.0)	0.0 (0.2)	1.7 (2.1)
<i>Populus trichocarpa</i>	0.0 (0.2)	0.0 (-)	0.5 (1.7)	0.0 (-)
<i>Prunus emarginata</i>	0.0 (-)	0.4 (0.9)	0.5 (1.2)	0.0 (-)
<i>Pseudotsuga menziesii</i>	2.3 (2.1)	3.8 (5.3)	3.5 (4.4)	6.0 (2.0)
<i>Rhamnus purshiana</i>	1.6 (2.9)	0.6 (1.1)	1.5 (2.2)	0.3 (0.6)
<i>Sorbus aucuparia</i>	0.4 (1.1)	0.1 (0.4)	0.0 (-)	0.0 (-)
<i>Taxus brevifolia</i>	0.0 (-)	0.1 (0.3)	0.2 (0.4)	0.0 (-)
<i>Thuja plicata</i> ^x	0.0 (-) ^a	1.4 (2.5) ^a	5.4 (5.5) ^b	11.3 (9.7) ^b
<i>Tsuga heterophylla</i> [*]	1.9 (3.7) ^a	0.3 (0.8) ^a	3.5 (4.4) ^b	12.7 (4.0) ^c
Number trees of all species ^x	21.4 (9.2) ^a	25.5 (12.0) ^{ab}	30.8 (9.7) ^b	35.0 (5.3) ^{ab}
# shade-tolerant trees	1.92 (3.74) ^a	1.75 (3.07) ^a	9.08 (8.25) ^b	24.00 (6.08) ^b
# conifers	6.6 (8.1) ^a	6.8 (6.9) ^a	13.7 (8.3) ^b	30.0 (4.4) ^b
# deciduous trees	14.8 (6.5) ^a	18.7 (8.5) ^a	17.1 (6.5) ^a	5.0 (3.6) ^b
# tree species	4.0 (1.2) ^a	4.7 (1.6) ^{ab}	5.6 (1.3) ^b	6.0 (-) ^{ab}
H'	1.06 (0.28) ^a	1.15 (0.26) ^{ab}	1.34 (0.23) ^b	1.29 (0.11) ^{ab}
Range species	2-6	2-8	3-8	5-5
n	24	24	24	3

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses; a superscript of "*" denotes that the natural log of the variable + 1 was used.

Table 5-10. Mean total diameter (dbh in cm) of trees per 500 m² (standard deviation) for the different sections. Only tree species present in at least two sections are reported.

Species	City	Mid	Far	OG
<i>Abies grandis</i>	18.2 (50.8)	10.2 (39.3)	8.7 (18.5)	0.0 (-)
<i>Acer macrophyllum</i> ^x	309.7 (159.2) ^a	295.3 (120.2) ^a	317.6 (136.9) ^a	64.4 (36.6) ^b
<i>Alnus rubra</i> ^x	80.4 (112.3) ^{ac}	150.0 (67.8) ^b	104.8 (103.7) ^a	0.0 (-) ^{ac}
<i>Amelanchier alnifolia</i>	0.3 (1.0)	1.3 (4.4)	0.0 (0.1)	0.0 (-)
<i>Chamaecyparis lawsoniana</i>	0.0 (-)	2.3 (1.5)	0.0 (-)	0.0 (-)
<i>Cornus nuttallii</i> ^x	4.1 (8.4) ^{ab}	2.0 (4.9) ^{ab}	0.3 (1.3) ^a	12.5 (12.6) ^b
<i>Populus trichocarpa</i>	1.9 (9.3)	0.0 (-)	13.7 (46.6)	0.0 (-)
<i>Prunus emarginata</i>	0.0 (-)	5.8 (14.8)	6.9 (15.9)	0.0 (-)
<i>Pseudotsuga menziesii</i> ^x	134.2 (126.8) ^a	127.7 (202.6) ^a	120.3 (131.0) ^a	644.4 (321.0) ^b
<i>Rhamnus purshiana</i>	2.9 (7.2)	0.9 (2.0)	1.2 (1.7)	0.2 (0.3)
<i>Sorbus aucuparia</i>	1.5 (6.6)	0.1 (0.7)	0.0 (-)	0.0 (-)
<i>Taxus brevifolia</i>	0.0 (-)	0.7 (3.3)	0.5 (1.2)	0.0 (-)
<i>Thuja plicata</i> ^x	0.0 (-) ^a	8.8 (14.7) ^a	80.8 (89.8) ^b	130.4 (71.6) ^b
<i>Tsuga heterophylla</i> ^x	33.3 (69.3) ^a	6.7 (27.6) ^{ab}	73.8 (98.8) ^{ac}	201.5 (112.5) ^d
dbh of all shade-tolerant trees ^x	33.3 (69.3) ^a	16.2 (33.7) ^a	155.0 (167.8) ^b	331.9 (66.3) ^b
dbh of all conifers ^x	185.7 (169.1) ^a	156.5 (205.7) ^{ab}	284.1 (191.4) ^{ac}	976.3 (300.1) ^d
Total dbh for all species ^x	586.6 (171.7) ^a	613.3 (198.7) ^a	728.8 (204.1) ^b	1053.4
(320.3) ^b				
n	24	24	24	3

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses.

Table 5-11. Mean tree diameter (dbh in cm) per 500 m² (standard deviation) for the different sections. Only tree species that were present in at least two sections and/or had an abundance of greater than 12 trees total are reported.

Species	City	Mid	Far	OG
<i>Abies grandis</i> n	8.0 (5.7) 60	15.3 (16.6) 16	9.3 (7.1) 32	0.0 (-) 0
<i>Acer macrophyllum</i> ^x n	26.9 (17.5) ^a 318	21.7 (17.1) ^b 364	22.3 (14.7) ^{ab} 415	19.1 (4.8) ^{ab} 11
<i>Alnus rubra</i> n	35.1 (7.5) ^a 61	28.3 (11.7) ^b 164	31.8 (9.9) ^{ab} 91	0.0 (-) ^{ab} 0
<i>Cornus nuttallii</i> n	8.6 (4.3) 10	6.0 (4.1) 8	5.2 (1.9) 2	7.5 (3.1) 5
<i>Prunus emarginata</i> n	16.2 (6.8) 3	13.9 (7.7) 28	17.0 (6.7) 29	0.0 (-) 0
<i>Pseudotsuga menziesii</i> [*] n	55.8 (31.3) ^a 64	32.4 (26.1) ^b 99	29.6 (23.2) ^b 108	80.7 (61.1) ^c 25
<i>Rhamnus purshiana</i> ^x n	2.1 (1.4) ^a 39	1.6 (1.4) 15	0.8 (0.5) ^b 40	0.5 (-) 1
<i>Thuja plicata</i> [*] n	0.0 (-) ^{ab} 0	7.3 (11.2) ^a 35	15.2 (16.6) ^b 155	11.6 (10.9) ^b 38
<i>Tsuga heterophylla</i> n	16.2 (18.9) 54	26.9 (25.9) 6	18.2 (17.0) 113	14.2 (15.6) 46
dbh for shade-tolerant trees [*] n	17.4 (20.3) ^a 46	9.3 (14.4) ^b 42	17.3 (17.9) ^a 218	13.8 (14.5) ^a 72
dbh for conifers n	28.0 (30.9) 159	22.8 (24.9) 164	20.8 (20.8) 329	32.5 (45.5) 90
dbh for deciduous trees n	27.2 (18.4) ^a 353	24.5 (16.4) ^{ab} 448	25.9 (15.8) ^{ab} 411	14.6 (8.1) ^b 15
dbh for all trees/quadrat n	27.4 (22.9) ^a 514	24.1 (19.0) ^{ab} 612	23.7 (18.4) ^b 740	29.9 (42.7) ^{ab} 105

Results of Bonnferoni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses; a superscript of "*" denotes that the natural log of the variable + 1 was used.

Table 5-12. Mean importance values (IV) of trees per 500 m² (standard deviation) in the different sections of the park. Only tree species that were present in at least two sections were reported.

Species	City	Mid	Far	OG
<i>Abies grandis</i>	0.09 (0.22)	0.05 (0.13)	0.04 (0.07)	0.00 (-)
<i>Acer macrophyllum</i> ^x	0.61 (0.28) ^a	0.75 (0.18) ^a	0.52 (0.20) ^a	0.08 (0.05) ^b
<i>Alnus rubra</i> ^x	0.26 (0.36) ^a	0.51 (0.29) ^b	0.26 (0.25) ^a	0.00 (-) ^a
<i>Amelanchier alnifolia</i>	0.01 (0.02)	0.01 (0.03)	0.00 (0.01)	0.00 (-)
<i>Cornus nuttallii</i>	0.03 (0.06)	0.01 (0.03)	0.00 (0.01)	0.06 (0.06)
<i>Populus trichocarpa</i>	0.01 (0.04)	0.00 (-)	0.03 (0.10)	0.00 (-)
<i>Prunus emarginata</i>	0.00 (-)	0.03 (0.07)	0.03 (0.08)	0.00 (-)
<i>Pseudotsuga menziesii</i>	0.37 (0.37)	0.30 (0.37)	0.28 (0.32)	0.76 (0.17)
<i>Rhamnus purshiana</i>	0.09 (0.12)	0.02 (0.03)	0.06 (0.10)	0.01 (0.02)
<i>Thuja plicata</i> ^x	0.00 (-) ^a	0.06 (0.10) ^a	0.29 (0.29) ^b	0.46 (0.35) ^b
<i>Tsuga heterophylla</i> ^x	0.13 (0.25) ^a	0.02 (0.07) ^a	0.19 (0.23) ^{ab}	0.56 (0.19) ^b
IV shade-tolerant ^x	0.13 (0.25) ^a	0.09 (0.15) ^a	0.48 (0.43) ^b	1.02 (0.25) ^b
IV conifers ^x	0.59 (0.46) ^a	0.45 (0.39) ^a	0.80 (0.38) ^a	1.79 (0.12) ^b
IV deciduous ^x	1.03 (0.53) ^a	1.33 (0.43) ^{ab}	0.91 (0.40) ^{ac}	0.15 (0.10) ^d
n	24	24	24	3

Results of Bonferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses.

value of conifers was significantly higher in the old-growth stand, as was the importance value of western hemlock. The importance value of deciduous trees, on the other hand, was greatest in the middle section of the park and lowest in the old-growth.

Shrub density, shrub species richness, shrub species diversity, and the total shrub basal area were all lowest at sites in the far section of the park (Table 5-13). Red elderberry, thimbleberry, and vine maple were the only shrub species to show significant differences among the sections. Red elderberry and thimbleberry were more abundant in the sections closer to the city, and vine maple was more abundant in the old growth stand. There were no significant differences in the distribution of species of non-native shrubs, but exotic shrub species were observed only at sites in the city and middle sections of the park.

Herbaceous species richness and diversity were greatest in the city section of the park (Table 5-14). Sites in the city section had significantly greater cover by species usually associated with shaded forest ecosystems such as fairy bells and star-flowered Solomon's seal. The lack of herbaceous covering (i.e., bare ground) was significantly higher at sites in the far section and the old-growth stand. Most of my sites had been cleared of ivy by Park personnel and volunteers before I collected my plant data. This made it impossible to adequately assess the problems and effects of ivy at my sites. The percent cover of non-native herbaceous plant species such as ivy could not be individually analyzed statistically due to the low numbers of plants and/or their being present at too few sites. Therefore, I analyzed the number and

Table 5-13. Mean shrub density, shrub species richness and diversity, and total shrub basal area (cm) and height (m) per 100 m² (standard deviation) for the different sections of the park. For scientific names see Appendix C.

Species	City	Mid	Far	OG
English laurel	0.0 (0.2)	0.0 (-)	0.0 (-)	0.0 (-)
Filbert	1.5 (3.7)	0.7 (1.3)	0.0 (-)	0.0 (-)
Himalayan blackberry	0.0 (0.2)	0.0 (-)	0.0 (-)	0.0 (-)
Holly	0.8 (1.9)	0.3 (1.0)	0.0 (-)	0.0 (-)
Indian plum	0.4 (0.9)	0.2 (0.6)	0.3 (0.5)	0.0 (-)
Mock orange	0.0 (0.2)	0.0 (-)	0.0 (-)	0.0 (-)
Oceanspray	0.2 (0.7)	0.8 (1.3)	0.2 (0.4)	0.3 (0.6)
Red elderberry ^x	3.4 (3.9) ^a	0.9 (1.4) ^b	0.6 (1.3) ^b	0.0 (-) ^b
Red huckleberry	2.4 (2.5)	2.0 (2.4)	1.4 (1.6)	3.7 (3.8)
Salmonberry	0.3 (1.4)	0.1 (0.3)	0.3 (1.1)	0.0 (-)
Snowberry	0.0 (0.2)	0.4 (1.3)	0.2 (0.7)	0.0 (-)
Thimbleberry ^x	1.4 (2.0) ^a	1.2 (1.7) ^a	0.1 (0.3) ^b	0.0 (-) ^{ab}
Vine maple ^x	0.8 (2.3) ^a	0.5 (1.4) ^a	0.8 (2.5) ^a	7.0 (10.4) ^b
Wild rose	0.5 (1.28)	0.8 (2.1)	0.5 (1.0)	0.0 (-)
Total # shrubs [*]	11.9 (6.3) ^a	7.9 (3.6) ^a	4.3 (4.0) ^b	11.0 (9.0) ^{ab}
# non-native shrubs ^x	0.9 (2.0)	0.3 (1.0)	0.0 (-)	0.0 (-)
# shrub species ^x	3.2 (1.7) ^a	3.3 (1.1) ^a	2.0 (1.3) ^b	2.0 (1.0) ^{ab}
Range of species per site	2-8	3-6	2-7	NA
H' ^x	0.86 (0.48) ^a	0.98 (0.33) ^a	0.49 (0.50) ^b	0.32 (0.39) ^{ab}
Total shrub basal area (cm) ^x	91.6 (63.1) ^a	81.2 (54.7) ^a	42.4 (51.7) ^b	20.1 (15.6) ^{ab}
Total shrub ht (m) ^x	7.8 (5.5) ^a	4.9 (2.1) ^a	2.7 (3.5) ^b	6.1 (4.8) ^{ab}
n	24	24	24	3

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses; a superscript of "*" denotes that the natural log of the variable + 1 was used.

Table 5-14. Mean percent cover by herbaceous species per 30 meter transect (standard deviation) for the different sections of the park. Only herbaceous species whose mean percent cover was greater than 1% (0.01) in at least one section were reported.

Species	City	Mid	Far	OG
Bedstraw	0.03 (0.04)	0.02 (0.03)	0.02 (0.04)	0.00 (0.00)
Bracken fern ^x	0.02 (0.03) ^a	0.01 (0.02) ^{ab}	0.01 (0.01) ^b	0.00 (-) ^{ab}
Candy flower	0.05 (0.06)	0.03 (0.03)	0.03 (0.08)	0.00 (-)
Fairy bells ^x	0.05 (0.05) ^a	0.01 (0.01) ^b	0.01 (0.01) ^b	0.00 (-) ^b
Fringe cup	0.00 (0.00)	0.01 (0.02)	0.02 (0.07)	0.00 (-)
English ivy	0.04 (0.11)	0.00 (0.00)	0.01 (0.04)	0.00 (-)
Lady fern	0.01 (0.02)	0.01 (0.02)	0.02 (0.06)	0.00 (-)
Oregon grape	0.11 (0.14)	0.12 (0.17)	0.10 (0.15)	0.23 (0.09)
Salal	0.01 (0.02)	0.02 (0.05)	0.02 (0.03)	0.03 (0.02)
Star-flowered S. seal ^x	0.05 (0.10) ^a	0.01 (0.03) ^b	0.00 (0.00) ^b	0.00 (-) ^{ab}
Star flower	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.02 (0.03)
Sword fern	0.26 (0.14)	0.38 (0.17)	0.36 (0.18)	0.13 (0.04)
Trillium	0.02 (0.01)	0.01 (0.01)	0.01 (0.01)	0.00 (0.00)
Inside-out flower	0.04 (0.08) ^{ab}	0.06 (0.08) ^a	0.01 (0.02) ^b	0.01 (0.01) ^{ab}
Yellow wood violet ^x	0.03 (0.04) ^a	0.01 (0.01) ^{ab}	0.01 (0.01) ^b	0.00 (-) ^{ab}
Water leaf	0.02 (0.03)	0.02 (0.04)	0.03 (0.03)	0.00 (-)
Wild blackberry	0.06 (0.05) ^a	0.12 (0.10) ^b	0.02 (0.03) ^a	0.01 (0.01) ^a
Wild ginger	0.02 (0.02)	0.01 (0.01)	0.00 (0.01)	0.00 (-)
Bare ground	0.13 (0.13) ^a	0.11 (0.05) ^a	0.28 (0.16) ^b	0.52 (0.16) ^b
Cover by non-native species	0.05 (0.11)	-0.01 (0.01)	0.01 (0.04)	0.00 (0.00)
# herbaceous species [*]	14.6 (3.2) ^a	12.3 (2.8) ^{ab}	10.3 (3.6) ^b	8.3 (2.3) ^b
Range of species	8-20	5-20	5-18	7-11
# non-native species ^x	1.2 (1.2) ^a	0.7 (0.9) ^{ab}	0.3 (0.6) ^b	0.7 (0.6) ^{ab}
# native species ^x	13.4 (3.6) ^a	11.6 (2.4) ^{ab}	10.0 (5.6) ^b	7.7 (2.1) ^b
H' (all herb species) ^x	1.98 (0.30) ^a	1.77 (0.32) ^{ab}	1.53 (0.38) ^b	1.35 (0.39) ^b
H' (native herb species only) ^x (0.38) ^{ab}	1.88 (0.31) ^a	1.73 (0.31) ^{ab}	1.49 (0.38) ^b	1.34
n	24	24	24	3

Results of Bonnferoni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses; a superscript of "*" denotes that the natural log of the variable + 1 was used.

percent of total non-native plants. I found significantly more non-native herbaceous plant species at sites in the city section.

There were some differences among the park sections regarding measurements associated with foliage height diversity (Table 5-15). There was significantly more vegetation (as measured by a reduced amount of sky) in the old-growth stand at two heights, and significantly less vegetation (more sky) in the middle section. There were also differences in the percent cover of certain tree species at different canopy heights, although these differences were often between the old growth stand and Forest Park.

Although there were no significant differences in the foliage height diversity (FHD) among the different park sections, there were differences with foliage diversity at different heights (Table 5-16). The old-growth stand had the greatest foliage diversity at heights of five, ten, and 50 meters, whereas the far section had the greatest foliage diversity at 20 meters, and the middle section had the greatest diversity at ground level.

DISCUSSION

Many plant community variables correlated strongly with one or more of the seven urbanization variables. I found significant differences in the density, richness, diversity, diameter, and importance values of trees, shrubs, and herbaceous plants in Forest Park and the old-growth stand along the defined urbanization gradient.

Table 5-15. Mean vegetation cover (as a proportion) for different species at different heights in the canopy are reported per 30 meter transect (standard deviation) for the different sections. Only species present in at least two sections and/or with an abundance of greater than 10 percent (0.10) in at least one park section are reported.

HEIGHT IN CANOPY	City	Mid	Far	OG
AT 0.5 METERS:				
Sword fern	0.40 (0.31)	0.56 (0.26)	0.54 (0.33)	0.40 (0.40)
Sky (no vegetation)	0.35 (0.32)	0.23 (0.19)	0.33 (0.29)	0.53 (0.31)
AT 2 METERS:				
Sky (no vegetation)	0.80 (0.21)	0.73 (0.16)	0.81 (0.22)	0.73 (0.31)
AT 3 METERS:				
Western red cedar	0.00 (-) ^a	0.00 (-) ^a	0.08 (0.14) ^b	0.17 (0.29) ^b
Western hemlock	0.08 (0.14)	0.01 (0.04)	0.07 (0.13)	0.20 (0.35)
Sky (no vegetation)	0.73 (0.21)	0.79 (0.19)	0.73 (0.22)	0.53 (0.42)
AT 5 METERS:				
Bigleaf maple	0.11 (0.18)	0.09 (0.16)	0.07 (0.11)	0.13 (0.23)
Western red cedar	0.00 (-) ^a	0.00 (-) ^a	0.06 (0.14) ^a	0.27 (0.12) ^b
Western hemlock	0.04 (0.10)	0.03 (0.09)	0.11 (0.19)	0.23 (0.06)
Sky (no vegetation)	0.80 (0.20) ^a	0.75 (0.25) ^a	0.69 (0.23) ^a	0.13 (0.23) ^b
AT 10 METERS:				
Red alder	0.03 (0.10) ^a	0.20 (0.24) ^b	0.01 (0.04) ^a	0.00 (-) ^{ab}
Bigleaf maple	0.31 (0.29)	0.30 (0.22)	0.19 (0.21)	0.13 (0.12)
Western red cedar	0.00 (-) ^a	0.00 (-) ^a	0.03 (0.07) ^a	0.20 (0.00) ^b
Douglas-fir	0.05 (0.08) ^{ab}	0.07 (0.14) ^{ab}	0.02 (0.06) ^a	0.20 (0.20) ^b
Western hemlock	0.04 (0.13)	0.02 (0.08)	0.06 (0.13)	0.13 (0.12)
Sky (no vegetation)	0.56 (0.32) ^{ab}	0.41 (0.30) ^a	0.68 (0.28) ^b	0.27 (0.31) ^{ab}
AT 20 METERS:				
Red alder	0.15 (0.20)	0.22 (0.29)	0.18 (0.21)	0.00 (-)
Bigleaf maple	0.60 (0.30) ^a	0.40 (0.34) ^{ab}	0.53 (0.24) ^a	0.00 (-) ^b
Douglas-fir	0.22 (0.29) ^a	0.14 (0.25) ^a	0.09 (0.19) ^a	0.70 (0.10) ^b
Western hemlock	0.01 (0.04) ^a	0.02 (0.08) ^a	0.08 (0.13) ^{ab}	0.23 (0.21) ^b
Sky (no vegetation)	0.01 (0.04) ^a	0.22 (0.35) ^b	0.05 (0.14) ^{ab}	0.07 (0.12) ^{ab}
AT 50 METERS:				
Douglas-fir ^x	0.25 (0.27) ^{ab}	0.13 (0.24) ^a	0.17 (0.29) ^{ab}	0.60 (0.20) ^b
Sky (no vegetation) ^x	0.74 (0.26) ^{ab}	0.88 (0.24) ^a	0.79 (0.31) ^a	0.33 (0.12) ^b
n	21	21	21	3

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses.

Table 5-16. Mean foliage diversity (FD) for different species at different heights in the canopy, mean foliage height diversity (FHD), and mean foliage height diversity without the ground layer (FHD-ground) per 30 meter transect (standard deviation) for the different sections.

Height in Canopy	City	Mid	Far	OG
FD at ground level	0.90 (0.30) ^{ab}	1.06 (0.30) ^a	0.74 (0.42) ^b	0.77 (0.47) ^{ab}
FD at 0.5 meters	0.53 (0.34)	0.58 (0.38)	0.38 (0.34)	0.29 (0.10)
FD at 2 meters	0.23 (0.23)	0.38 (0.23)	0.27 (0.27)	0.34 (0.34)
FD at 3 meters	0.37 (0.29)	0.31 (0.27)	0.32 (0.24)	0.34 (0.36)
FD at 5 meters	0.24 (0.23) ^a	0.31 (0.30) ^a	0.37 (0.27) ^a	1.10 (0.21) ^b
FD at 10 meters	0.37 (0.22) ^a	0.53 (0.25) ^a	0.38 (0.31) ^a	1.09 (0.49) ^b
FD at 20 meters	0.54 (0.30) ^{ab}	0.44 (0.32) ^a	0.68 (0.29) ^b	0.49 (0.27) ^{ab}
FD at 50 meters	0.20 (0.16) ^{ab}	0.08 (0.15) ^a	0.14 (0.21) ^{ab}	0.39 (0.27) ^b
Total FHD	3.37 (0.98)	3.28 (1.09)	3.69 (0.98)	4.80 (1.10)
FHD - ground	2.47 (1.02) ^a	2.54 (0.97) ^a	2.63 (0.98) ^a	4.03 (1.15) ^b

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

The distance from an urban center, probably the simplest method of describing an urbanization gradient, implies that sites nearer to the city are the most urbanized. Using this single measurement as a gradient, I found lower tree density, tree species richness, tree species diversity, and importance of shade-tolerant trees, as well as smaller total tree diameter (total dbh) at more urban sites. Measurements involving shade-tolerant tree species were frequently negatively correlated with urbanization. I found lower densities and importances of shade-tolerant trees and conifers, as well as lower densities of such shade-tolerant coniferous species as western hemlock and western red cedar at more urban sites.

Many measures of shrubs were negatively correlated with the distance from Portland. I found decreased density and species diversity of shrubs, as well as lower total height of all shrub species with increased distance from Portland. Density and height of thimbleberry and red elderberry also decreased with distance from downtown. Vine maple was the only shrub species that was positively correlated with distance from Portland as evidenced by lower density, height, basal area, and importance at more urban sites.

Herbaceous cover generally decreased as distance from Portland increased. I found larger numbers of herbaceous species, both native and non-native, as well as greater species diversity for all herbaceous species and for native herbaceous species at more urban sites.

Although my contention is that urbanization in the Portland area is causing the measured effects on the plant community of Forest Park and the surrounding forested

areas, it is also possible that there are other causes for the observed results. One possible explanation that excludes direct effects of humans is the peninsula effect. The peninsula effect predicts greater numbers of species in the old-growth (the "mainland" end), with decreasing numbers of species at sites closer to Portland. However, sites in the far section of Forest Park had more tree species and greater tree species diversity than the old-growth, sites in the city and middle sections had more shrub species and greater shrub species diversity than sites in the far section of Forest Park, and sites in the city section had more herbaceous species and higher herbaceous species diversity than sites in either the far section or the old-growth in contrast to what the peninsula effect would predict. My data are not consistent with a peninsula effect. It appears, rather, that environmental parameters (e.g., temperature or moisture at a site) or temporal factors (i.e., the length of time since the last major disturbance at a site) probably have greater roles in species distribution in Forest Park than does a peninsula effect. This conclusion is also supported by the findings of Milne and Forman (1986) who determined that environmental heterogeneity had a larger impact on species diversity than did a peninsula effect.

It is unlikely that "natural" environmental parameters are responsible for the observed results because Forest Park is relatively small, and environmental and edaphic factors are similar throughout the study area. In any case, variation due to such factors would be accounted for in the regression coefficients. It seems more likely that different sections of the park are in different stages of succession. In particular, changes in the distribution of shade-tolerant tree species (associated with later successional stages) relative to that of shade-intolerant species (associated with earlier successional stages) suggest that succession is very important in defining site

characteristics. These changes indicate that continual and/or increased disturbance related to urbanization maintains more urbanized sites at earlier successional stages than at less urbanized sites.

My results suggest that the four sections studied (i.e., the three sections of Forest Park and the old-growth stand) are in different successional stages. The old-growth stand is significantly different from sites in Forest Park, probably because continual human disturbances are more prevalent in Forest Park than in the, until recently, relatively undisturbed old-growth stand. In the old-growth stand, I found greater abundance and dominance of shade-tolerant tree species, as well as trees of larger diameter than at any site in Forest Park. These observations confirm the idea that the old-growth stand is in a mature or later successional stage of forest development than any site in Forest Park. In addition, most measurements of canopy cover showed that the old-growth was significantly different from sites in Forest Park. Foliage height diversity was not significantly different for the different sections studied, although it was significantly higher in the old-growth when herbaceous ground cover was removed. This is in keeping with results found by Portland Parks and Recreation (1995) showing that the park in general has reduced vertical structure, a characteristic of earlier successional stages in forest development (Spies 1991; Wilson 1991).

Sites in the middle section were dominated by deciduous trees, particularly red alder. In addition, the trees in the middle section had smaller diameters than those in the other sections. Red alder is an early successional species in the Pacific Northwest (Franklin and Pechanec 1967; Houle 1996; Newton et al. 1967). Alders usually

invade and dominate sites where soils have been deeply scarified to the exclusion of Douglas-fir (Franklin and Pechanec 1967; Newton et al. 1967). Available information confirms that much of this section was burned in 1951 (Bill Keil, retired forester for Portland Parks and Recreation, personal communication, Dec. 1997). According to Keil, only two sites in this section (sites 18 and 32) were outside of the boundaries of that fire. Site 32 was similar in species composition to the other sites in the middle section, and there was evidence of fire at the site in the form of charred stumps. This site may have escaped the fire of 1951, but it had burned at some recent point in its history. If Keil is correct, it was probably logged and slash burned at some time prior to 1951. Site 18 was very different from the other sites in the middle section. Site 18 had the most tree species (nine) of any site in that section of the park (mean number of species was 4.7). Among the species present was Pacific yew, a species associated with older successional forests (Spies 1991). Site 18 is the only site west of the area burned by the 1951 fire that is not in the far section. It appears to be in a successional stage that is more similar to sites in the far section than to sites in the middle section.

The far section of Forest Park, characterized by sites having increased dominance and importance of shade-tolerant tree species, seems to be in a later successional stage than other sections of the park. The distribution of Pacific yew supports this view, although Pacific yew was not present at enough sites or in sufficient numbers to analyze statistically. The species was found only at sites in the middle (site 18) and far (sites 6, 7, and 22) sections of the park. Although Pacific yews are most frequently found in older successional stands, their distribution is more influenced by past land use history and available seed sources than by seral stage (Busing et al.

1995; Spies 1991). The increased abundance of shade-tolerant tree species also supports the contention that the far section is older than the middle or city sections.

The city section of Forest Park was difficult to place in a successional sequence relative to the far section. Sites in the city section had trees with larger diameters than did other sections of Forest Park (e.g., the city section had larger Douglas-firs, shade-tolerant trees, shade-intolerant trees, and deciduous trees than the middle section; it also had larger Douglas-firs and bigleaf maples than the far section). The increased diameter of trees relative to the middle section leads me to conclude that the city section is in a later seral stage than the middle section of the park. In addition, the fact that trees in general, and Douglas-firs in particular, were larger in the city section than in the far section suggests that perhaps sites in the city section are in the oldest successional stage in Forest Park. This interpretation is questionable, however, because of the increased dominance of shade-tolerant trees in the far section. The evidence suggests that some trees in the city section may be older than trees in the far section, but that factors not considered here may be causing aspects of the plant community in the city section to exhibit characteristics of an earlier successional stage.

There appears to be a successional gradient in Forest Park due to the effects of logging and/or fires. However, that successional gradient does not imply that there has been no additional human affect on the distribution of trees and other plant species in the park. It may be that human disturbance is altering the "normal" patterns of succession in the city section of Forest Park similar to what has occurred in other disturbed forests (Abrams and Nowacki 1992; Dale and Doyle 1987; Hong

et al. 1995; Kowarik 1990; Matlack 1993b). As areas surrounding Forest Park have been developed, the park has become increasingly fragmented (Portland Parks and Recreation 1995). Such fragmentation results in an increased amount of edge habitat. In my results, a greater amount of edge habitat would be indicated by a dependent variable exhibiting negative correlations with the distance from the nearest forest edge, the distance from the nearest road, and the percent of surrounding forest, as well as positive correlations with the distance from the center of the forest and the number of surrounding houses. Tree species density, richness, and diversity, as well as density of conifers and shade-tolerant tree species, were positively correlated with edge habitat. The fact that most tree variables were associated with edge habitat lends credence to the theory that edge resulting from habitat fragmentation, a consequence of human disturbance, has had dramatic impacts on plant community structure in Forest Park just as has been reported for other areas (Airola and Buchholz 1984; Dzwonko and Loster 1988; Greller et al. 1990; Rudnický and McDonnell 1989). Human disturbance frequently results in plant communities that exhibit characteristics of earlier seral stages (Abrams and Nowacki 1992; Dale and Doyle 1987; Hong et al. 1995; Kowarik 1990; Leak and Smith 1996; Matlack 1993b; Petraitis et al. 1989; Swindel et al. 1986). The successional stages I have found indicate that humans have had and will probably continue to have a dramatic impact on the structure of Forest Park. As human disturbance, especially in the form of increased habitat fragmentation, increasingly affects Forest Park, it would be expected that plant community structure in Forest Park will continue to be affected.

Examination of the distribution of shrubs and herbaceous plants also supports the idea of a successional gradient influenced by humans. The old-growth stand had

more vine maple than any site in Forest Park and, along with the far section, it had a higher percentage of bare ground. Spies (1991) found significantly more vine maple at older successional sites located along the Oregon coast. The increased occurrence of vine maple in the old-growth site supports the idea that this site is in a mature stage of forest succession. Spies and Franklin (1991) found that, in general, herbaceous cover increased with stand age class. However, they also pointed out that understory species varied widely depending on the overstory composition, site moisture, and site disturbance history. More than likely, decreased herbaceous cover in the far section of the park and the old-growth stand can be attributed to the increased density of trees and saplings, and to decreased soil moisture at the location of the transects since I noticed that lower, wetter sites had a higher coverage of herbaceous plants. In addition, the old-growth stand contained such old-growth associates as *Corallorhiza* sp. (coral-root orchid), *Tiarella trifoliata* (foam flower), and *Linnaea borealis* (twinner) (Ruggiero et al. 1991; Spies 1991).

The city and middle sections had greater shrub species diversity and shrub species richness than did the far section. The city section also had a greater density of shrubs than the middle or far sections. The greater density of shrubs at more urban sites would be expected in earlier successional states, as well as in response to continual disturbance (Carey et al. 1991; Norse 1990). Fugate (1994) found positive correlations between shrub species richness and the distance from the edge of the forest. Although I did not find the same correlations in this study as Fugate, I did find positive correlations of other shrub variables with edge habitat. For example, the densities of non-native shrubs and vine maple plants were found to be related with edge habitat. The density of non-native shrubs correlated positively with the

number of houses. Since many exotic species escape from gardens, the positive correlation with houses is not unexpected. Thus, in the shrub community, urbanization is correlated with an increased abundance of non-native species.

Whereas many shrub variables were associated with edge habitat, many herbaceous variables were associated with forest interior. Increased herbaceous species diversity and greater cover of several native herbaceous species were associated with forest interior habitat. The increase in herbaceous species near the city could be the consequence of historical usage because the city section appears to have been sheltered from fires and logging for longer than the other sections of the park. The city section has only experienced one small salvage logging job since 1951 that followed the Columbus Day storm in 1962 (Keil, W., retired forester for Portland Parks and Recreation, personal communication, Dec. 1997). Only one site (14) in the city section provided physical evidence of logging or fire, whereas six sites in the far section (all except site 7) provided evidence of such human disturbance (Table 5-17).

The distribution of non-native plant species followed a predictable pattern. Non-native tree and shrub species were present only at sites in the city and middle sections of Forest Park. Non-native herbaceous plants were present at sites in all sections, but there were significantly more non-native herbaceous species present at sites in the city section than at sites in the far section. There were too few non-native trees to be statistically analyzed by either regression or ANOVA. However, increased density of non-native shrubs and increased cover by non-native herbaceous species were both associated with increased edge habitat. Since increased species richness and

Table 5-17. Types of disturbance that have occurred in the past as determined by physical evidence of fire and/or logging activity observed at each site.

City Section:

<u>Site #</u>	<u>Evidence at Site</u>
1	-
3	-
5	-
12	-
13	-
14	logged
28	-
30	-

Middle Section:

<u>Site #</u>	<u>Evidence at Site</u>
11	logged/fire
17	-
18	-
23	-
24	-
32	fire
36	logged/fire
37	logged/fire

Far Section:

<u>Site #</u>	<u>Evidence at Site</u>
2	fire
6	logged/fire
7	-
15	logged
19	logged/fire
22	fire
39	logged/fire
41	fire

Old-growth:

<u>Site #</u>	<u>Evidence at Site</u>
OG	fire

species diversity with proximity to edge is well documented (Chen et al. 1992, 1995; Fugate 1994; Lehmkuhl and Ruggiero 1991; Levenson 1981; Matlack 1993a; Ranney et al. 1981), my results are not surprising.

Increased tree species richness and diversity were associated with edge habitat in this study, but both were significantly greater in the far section and the old-growth stand. These results suggest that although edge effect may be important to the distribution of trees in Forest Park, the past land use and history successional stage of the site is probably also very important. Increased species diversity of both shrubs and herbaceous plants was associated with forest interior habitat, although herbaceous species richness and diversity were significantly greater at sites in the city section. This result did not depend on including non-native species. Thus, the increase in species diversity was not merely the result of an influx of exotic species. The intermediate disturbance hypothesis may be the best explanation for the increased species diversity of herbaceous plants at the city end of the park. According to this hypothesis, moderate levels of disturbance lead to increased species diversity due to increased heterogeneity of disturbed habitat versus nondisturbed habitat (Connell 1978; Huston 1979; Petraitis et al. 1989). Disturbances can be described in terms of both severity and frequency. Intermediate levels of repeated disturbances help maintain the patchiness of a landscape, affording a greater variety of habitat types and allowing for a more heterogeneous distribution of species (Petraitis et al. 1989). A number of investigators have reported that plant species diversity initially increased and then decreased as disturbance intensified (Armesto and Pickett 1985; Hoehne 1981; Kowarik 1990), similar to the findings in this study.

The level of disturbance in any forest should correlate with edge. The city section of Forest Park has significantly less area of forest interior habitat than other sections as a result of development and habitat fragmentation (Figure 5-1). Concomitantly, the city section has a greater amount of edge habitat (Figure 5-2). Increased edge habitat implies increased levels of disturbance. If this correlation is correct, the increased diversity at sites in the city section is to be expected. The diversity of all herbaceous species and many species of native herbaceous plants correlated positively with the distance from the nearest road suggesting that disturbance associated with roads may be causing the observed reduction in species diversity at sites near roads. This correlation could be due to the increased trampling (increased disturbance) near roads in the park, as roads provide access points for entry (Matlack 1993b). High levels of continual disturbance could lead to increased homogeneity and thereby to decreased species diversity at sites near roads, whereas areas further from roads would have less trampling and less disturbance even though they may still be more highly disturbed than other areas of the park. The city section of Forest Park has significantly higher usage by humans than other sections (Figure 5-3) due in part to a higher density of trails (Figure 5-4; adapted from data from Portland Parks and Recreation 1995) and probably more convenient access. The potential for physical disturbance and/or trampling of vegetation at a given site is therefore greater at more urban sites. The results may support the intermediate disturbance hypothesis where sites in the more highly utilized city section have increased species diversity due to moderate levels of disturbance, whereas areas nearer to roads have decreased species diversity as the result of higher and/or more continual levels of disturbance.

Figure 5-1. The area (ha.) of forest interior habitat in the different sections of the park assuming that edge extends 250 or 500 meters into the forest.

Figure 5-2. The percent of forest edge habitat in the different sections of Forest Park assuming that edge extends 250 or 500 meters into the forest.

Figure 5-3. Mean site usage (total human usage) in different areas of the park.

Figure 5-4. The density of trails (miles per acre) in the different sections of Forest Park.

Assuming that the old-growth stand more closely resembles the original native forest than other younger forested lands, the forest that I found growing in Forest Park was very different from the presettlement forest. I believe that this is the result of continual and dramatic disturbance events -- logging and forest fires -- that have resulted in a mosaic of landscapes. Many of my results can be explained by the observation that the different portions of the park are in different successional stages. However, humans appear to be a moderating influence on the rate of succession. Their effects are especially apparent in the city section of the park and may be retarding successional changes in the city section of Forest Park.

CHAPTER 6

THE EFFECTS OF HUMAN DISTURBANCE AND URBANIZATION ON THE POPULATION DYNAMICS AND VIGOR OF TREES IN THE FOREST

Disturbance can cause shifts in the demographic structure of a forest, depending upon the degree and frequency of the disturbance (Harper 1977). Rao et al. (1990) reported that tree species in disturbed, subtropical forests in India exhibited reverse-J size distribution curves (i.e., an age structure showing large numbers of individuals in the smallest size class with a steep decline and flattening among larger size classes), whereas those in undisturbed forests exhibited either sigmoidal or bimodal curves. If urbanization increases the amount of disturbance in a system, then urbanization should have significant effects on the age structure of trees in a forest. Rudnický and McDonnell (1989) are among the few researchers who have addressed the effects of urbanization on the demography of trees in a natural forest system. They found that the diameter size class distribution of trees in the New York City Botanic Garden Forest had changed over a 48-year period. Their research revealed a large increase in the number of smaller stems, especially for certain tree species. They also found a reverse-J curve for the diameter size class distribution of all species of trees in New York City. However, the same reverse-J size distribution that indicates disturbance in some forests is reported to be normal for old-growth forests in the Pacific Northwest (Arsenault and Bradfield 1995; Franklin and Spies 1991b; Parker and Peet 1984). Urbanization, a type of disturbance, should affect the population dynamics of nearby forests. If older age classes are damaged or

destroyed, the proportion of seedlings and saplings should increase, although if soil microenvironments in urban ecosystems are severely disrupted by soil compaction or the effects of pollutants, there should be a decrease in the proportion of seedlings and saplings (Airola and Buchholz 1984; Greller et al. 1990; Kowarik 1990; Pouyat et al. 1994, 1995a, 1995b; Rogers and Rowntree 1988; Seidling 1990; Supuka 1994; Vanhala et al. 1996; White and McDonnell 1988; Yost et al. 1991).

Rudnicky and McDonnell (1989) found in their study an apparent lack of reproduction of certain native tree species in the New York City Botanic Garden Forest, especially for *Tsuga canadensis* and species of *Quercus*. Similar reproductive failures have also been reported at other disturbed locations (Airola and Buchholz 1984; Greller 1975; Hoehne 1981; Levenson 1981; McBride and Jacobs 1986; Yost et al. 1991). James et al. (1979) found a reduction in the abundance of tree seedlings in campgrounds in Ontario. Airola and Buchholz (1984) found decreased reproduction by shade-tolerant tree species (e.g., *Acer rubrum* and *Quercus alba*), but increased reproduction by early seral ("weedy") tree species (e.g., *Ailanthus altissima* and *Prunus serotina*) in more disturbed areas. Yost et al. (1991) noticed a reduction in the abundance of tree seedlings as the percentage of non-native herbaceous species increased. Proposed causes for the observed decline in tree reproduction include the effects of trampling and arson, increased browsing by small mammals such as squirrels, competition by non-native plants, and the effects of pollution. Many scientists have reported reductions in the abundance of seedlings of native, dominant tree species in areas with high concentrations of atmospheric pollutants (Greller et al. 1990; Klein and Perkins 1988; Kozlowski et al. 1991; Rudnicky and McDonnell 1989; Woodward 1992).

Abiotic factors related to urbanization, such as increased amounts of air pollution or soil compaction, could be expected to cause reduced tree vigor and/or increased mortality (Airola and Buchholz 1984; Greller et al. 1990; James et al. 1979; Klein and Perkins 1988; Kowarik 1990; Pouyat et al. 1994, 1995a, 1995b; Rogers and Rowntree 1988; Rudnický and McDonnell 1989; Seidling 1990; Supuka 1994; Vanhala et al. 1996; White and McDonnell 1988; Yost et al. 1991). Tree vigor relates to the overall health of the tree and is usually measured as diameter increment growth (Ashby and Fritts 1972; Fitzgerald and Raynal 1991; James et al. 1979; Jones et al. 1994; McBride and Jacobs 1986; McLaughlin et al. 1983; Nash et al. 1975; Rudnický and McDonnell 1989; Thompson 1981). However, other criteria such as crown fullness, amount of twig dieback, needle loss (Fitzgerald and Raynal 1991), and the degree of regeneration (Jones, et al. 1994; McBride and Jacobs 1986) are also employed. James et al. (1979) found reductions in both annual increment growth and needle length in areas of heavy human disturbance. Several researchers found reductions in the size of annual rings in areas with moderate to high levels of air pollution (Ashby and Fritts 1972; McLaughlin et al. 1983; Nash et al. 1975; Thompson 1981). Continued reductions in tree vigor could be expected to be correlated with increased tree mortality. McBride and Jacobs (1986) found reduced radial increment growth along with increased tree mortality in some California urban areas. Urbanization has also been reported to cause increases in the mortality rates of certain tree species (Levenson 1981, McBride and Jacobs 1986, Rudnický and McDonnell 1989). Increased concentrations of heavy metals, such as lead, that accumulate in urban soils have been correlated with increased tree mortality rates (Airola and Buchholz 1984; Pouyat et al. 1994; Pouyat et al. 1995a, 1995b; Supuka

1994; White and McDonnell 1988; Yost et al. 1991). If mortality is random, there should be no species or age-specific differences in the death pattern, and there will be no apparent effect on the age structure of the forest. If, however, mortality is more common in certain species or if it is more prevalent at a specific portion of the age structure (i.e., very old or very young trees), the structure of the forest will change.

Many investigators have argued that changes occur in the age structure of a forest along an urbanization gradient (Airola and Buchholz 1984; Greller 1975; Hoehne 1981; Levenson 1981; McBride and Jacobs 1986; Rudnický and McDonnell 1989; Yost et al. 1991). These changes are often manifested by differential rates of either regeneration or mortality or both. In addition, the health or vigor of trees also seems to be affected by the processes of urbanization (Airola and Buchholz 1984; Greller et al. 1990; Klein and Perkins 1988; Kowarik 1990; Pouyat et al. 1994, 1995a, 1995b; Rogers and Rowntree 1988; Rudnický and McDonnell 1989; Seidling 1990; Supuka 1994; Vanhala et al. 1996; White and McDonnell 1988; Yost et al. 1991). In this study, I have tried to determine if changes are occurring in the age structure of trees in Forest Park along an urbanization gradient and I have examined tree regeneration, tree mortality, and the vigor of trees as determined by increment growth.

METHODS FOR POPULATION DYNAMICS

The diameters at breast height (dbh) of all living trees and saplings in each 500 square meter quadrat were measured using a diameter tape. Each tree or sapling was placed into a size class based upon its dbh. There were two classes of saplings: saplings 0 - 4.9 cm dbh and saplings 5.0 - 9.9 cm dbh. There were four categories

of trees: trees with dbh 10.0 - 19.9 cm; 20.0 - 39.9 cm, 40.0 - 59.9 cm, and greater than 60.0 cm.

Randomly selected trees at each site were cored at a height of approximately one meter. This height was chosen because it was lower to the ground than breast height, and hence would provide more accurate tree ages, while still allowing for the leverage necessary to core a hardwood tree. Sites with many trees were sampled by randomly selecting trees from several size classes for all species present. The cores were sanded and stained using phloroglucinol. The number of rings was counted under a dissecting microscope to estimate the age of the tree. To determine the ages of trees too small to core (<5 cm dbh), randomly selected saplings were harvested, a stem disk was taken from near-ground levels, and the rings counted on the round obtained.

The increment cores were also used to determine whether the tree's vigor, as revealed by growth rate, had changed over time or along a spatial urbanization gradient over the last 70 years. Growth rates were measured as the average width of annual rings measured in decade intervals. Decade intervals were used to minimize annual climatic fluctuations. This analysis was done for seven consecutive decades on all cores where it was possible and for as many decades as possible on trees with fewer than 70 annual rings. In addition, the live crown ratio (the ratio of the height of the lowest living branch to the height of the tree) was used as another means to compare the vigor of trees among sites.

Regeneration and mortality of trees were determined at each site. Regeneration was measured by counting the number of living seedlings of each tree species located along each transect and by counting the number of living saplings in each quadrat. All dead trees located in each quadrat were counted, their height and dbh were measured, and the trees were identified to species when possible. These mortality measurements were reported for both standing and recently fallen trees. Sites were visited for four consecutive years (1993 through 1996), tree mortality was recorded annually, and a rate of mortality was calculated for each location.

RESULTS

Data Analysis

The dependent variables that were related to tree demographics were divided into related groups: tree regeneration, age structure, tree mortality, and tree growth rates. As before, all dependent variables were analyzed using multiple regression and ANOVA. Each dependent variable was analyzed by multiple regression using the seven independent urbanization variables and the covariable as discussed previously. Each significant dependent variable was further examined using ANOVA, and the various sections of the park were compared using Bonferroni/Dunn analysis to detect significant differences. Regression coefficients are located in Appendix E.

Multiple Regression Analysis of Tree Demographics -- Age Structure, Recruitment, Mortality, and Growth Rates

Several dependent variables associated with the demographics of trees and saplings at sites in Forest Park and the old-growth stand were found to be correlated with at least one urbanization variable by multiple regression (Table 6-1). Variables that were

Table 6-1. Results of multiple regression analysis for tree population structure -- tree abundance by size class. Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). n=75 except where noted. For regression coefficients, see Appendix E.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 †
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	
sqrt(sapl. 5-10)	++				++		-	0.32***
sqrt (conifer sapl. <5)	+			--				0.32***
sqrt (conifer sapl. 5-10)	+++							0.32***
sqrt (shade-tol. <5cm)	+++					-		0.34***
sqrt (shade-tol. 5-10cm)	+++	-						0.37***
sqrt (shade-tol. 10-20cm)	+	-						0.25**
sqrt (shade-tol. 40-60cm)	++							0.33***
sqrt (shade-intol. 10-20cm)	+							0.25*
sqrt (shade-intol. >60cm)	--					+		0.28**
sqrt (alders 20-40cm)				--		-		0.27**
sqrt (cascara sapl. <5cm)			--				+	0.34***
sqrt (hemlock sapl. <5cm)				+				0.38***
sqrt (w.r.cedar sapl. <5cm)	+++			---		--		0.40***
sqrt (w.r.cedar sapl. 5-10cm)	+++							0.31***
sqrt (w.r.cedar 10-20cm)	+++	---				--	-	0.49***
sqrt (mean age Douglas-fir)‡			+					0.25*

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

‡ n=60

positively correlated with the distance from downtown Portland included the density of larger (5-9.9 cm dbh) saplings, and the densities of shade-tolerant trees and coniferous trees. The only variable that was negatively correlated with this urbanization variable was the density of larger shade-intolerant trees. Other urbanization variables that were correlated with tree population structure included the distance from the nearest road (negatively), the distance from the nearest edge (mostly negatively), and the percent of surrounding forest (mostly negatively).

Several independent variables that measured tree mortality were also positively correlated with the distance from downtown Portland (Table 6-2). The numbers of dead western hemlocks, dead western red cedars, dead conifers, dead shade-tolerant trees, and dead trees of all species were each positively correlated with the distance from downtown Portland, as was the percentage of trees on a site that were dead in 1996. Three of these same variables were also positively correlated with the distance from the nearest forest edge.

Tree increment growth rates (GR) were found to be correlated with urbanization variables for several species (Table 6-3). Growth rates of coniferous species such as western red cedar, grand fir, and western hemlock for the last 4 decades were often positively correlated with the distance from the nearest road. The number of surrounding houses, the distance from the nearest forest edge and the percent of surrounding forest were also significant in many cases. Douglas-fir growth rates for earlier decades were negatively correlated with the distance from the nearest forest edge and positively correlated with the percent of surrounding forest.

Table 6-2. Results of multiple regression analysis for tree mortality. Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix E.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 †
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	
sqrt (dead hemlocks)	++			+		+		0.38***
sqrt (dead w.r. cedar)	+++	--	-		+		---	0.48***
sqrt (# dead conifers)	++			+			-	0.34***
sqrt (# shade tolerant)	+++	--						0.43***
sqrt (# dead trees)	+++							0.26**
% of dead trees in 1996	+			+				0.25*

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

Table 6-3. Results of multiple regression analysis for tree increment growth rates (GR) as mean cm/year for each decade (number in parentheses refer to decade). Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). All independent variables are square root transformations. n varies and is noted. For regression coefficients, see Appendix E.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 [†]	n
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER		
GR 10-20 cedar	+		-					0.26*	55
GR 20-30 cedar					+		-	0.50***	49
GR 10-20 grand fir [‡]		+++	--	+	+++		--	0.61**	30
GR 20-30 grand fir [‡]		++						0.69**	23
GR 20-30 hemlock	+	++						0.33**	61
GR 30-40 hemlock		+						0.38**	52
GR 40-50 Douglas- fir		-		-		+	-	0.36***	99
GR 50-60 Douglas- fir				-		+		0.31***	91
GR 60-70 Douglas-fir				-		+		0.35***	85
GR 50-60 maple		+++			+		--	0.32**	84
GR 50-60 alder [‡]		+		+		--		0.33*	42

[†] Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

[‡] cannot be analyzed with regression while using covariable -- tree was not present in the old growth

The live crown ratio (LCR) was only significant for four dependent variables (Table 6-4). Generally, the live crown ratio was negatively correlated with the distance from downtown Portland and the distance from the nearest forest edge, and positively correlated with the distance from the nearest road.

Tree Demographics -- Age Structure, Recruitment, Mortality, and Growth Rates -- Differences Between Sections

Age Structure

There were no differences observed in the distribution of seedlings as evaluated by either regression or ANOVA. Indeed, there were relatively few seedlings measured at any site (Table 6-5). There were no significant differences in the numbers of seedlings among the different sections, and although the middle section had 82 bigleaf maple seedlings, all but four were counted on one transect. Observations later the same year found that none of these seedlings had survived. The far section was the only area where seedlings of any coniferous species (western red cedar) were counted.

Sites in the city and middle sections had significantly fewer shade-tolerant saplings and trees of all sizes than sites in the far section or the old-growth (Figure 6-1). There were significantly more shade-intolerant small saplings (<5 cm dbh) in the middle section than the other sections. There were also significantly more shade-intolerant trees in the 10-20 cm size class in the middle and far sections of the park. There were significantly more very large (>60cm dbh) trees in the old-growth. For ANOVA and Bonnferoni/Dunn analysis results, see Appendix F.

Table 6-4. Results of multiple regression analysis for live canopy ratio (LCR) in upper canopy trees (up. can.). Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). Sample size (n) varies and is noted for each variable. For regression coefficients, see Appendix E.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 [†]	n
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER		
LCR grand fir ‡	---	+++	--	+	+			0.30***	108
LCR up can b.l.maple	---	+++	+++	---	-		+	0.31***	204
LCR up can Douglas-fir		++		-			-	0.39***	148
LCR up can intol trees	---	+++	+++	---	--		+	0.32***	463

[†] Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

‡ cannot be analyzed with regression while using covariable -- tree was not present in the old growth

Table 6-5. Total number by species of seedlings found in the different sections of the park.

Species	City	Middle	Far	OG
<i>Acer macrophyllum</i>	0	82 [†]	2	0
<i>Rhamnus purshiana</i>	8	0	3	1
<i>Thuja plicata</i>	0	0	4	0
<i>Prunus emarginata</i>	0	0	2	0
Total # seedlings	8	82*	11	1
Mean # seedlings (standard deviation)	0.3 (0.9)	3.4 (15.9)	0.5 (1.0)	0.3 (0.6)
# species	1	1	4	1

[†] Seedlings on one transect at this site were counted in June 1995. All other transects were counted between July 1 and August 31, 1994. Observations later in the summer of 1995 revealed no living seedlings on this transect.

Figure 6-1. Size class distribution of trees in the different sections of the study. Each bar shows the percent of shade-tolerant versus shade-intolerant species.

The mean age of all trees whose cores I could age (i.e., generally trees less than 50 cm dbh) was greatest for the site in the old-growth (62 years), followed by the city section (51.5 years), the far section (43.7 years), and then the middle section (38.8 years) (Table 6-6). Trees, on average, were significantly older at sites in the old-growth and the city section than at sites in the middle and far sections of Forest Park. Bigleaf maples were significantly older in the city than in the middle or far sections, and alders and Douglas-firs were older in the city than in the middle section.

Mortality

There were significantly more dead trees in the far section than in either the city or middle section of Forest Park (Table 6-7). There were more dead conifers and shade-tolerant trees in the far section and the old-growth. There was significantly more dead bigleaf maple in the far section, more dead red alder in the middle section, and more dead western hemlock and western red cedar in both the far section and the old-growth.

Increment Growth Rate and Live Crown Ratio

For all decades examined, the average increment growth rates (cm/year) for both Douglas-firs and bigleaf maples were lowest in the old-growth stand (Figure 6-2). Growth rates for trees in Forest Park had a unimodal pattern where rate of growth had peaked between 30 and 50 years ago with slower rates before and after that time. The growth rates for the two most recent decades (0-10 and 10-20 years ago) tended to be significantly lower than the other decades for all sections of the park. The old-growth (outside of the park) had a very different pattern with no significant pattern

Table 6-6. Mean age of trees that were small enough to be aged by coring (standard deviation) for the different sections. Tree species that were present in less than two sections were omitted from the table. Reported results are for only trees for which complete core samples were available (generally <50 cm dbh). n varies by section and species and is reported for each variable.

	City	Mid	Far	OG
Bigleaf maple ^x	57.4 (28.4) ^a 50	43.1 (15.1) ^b 59	44.0 (13.2) ^b 49	62.2 (16.8) ^{ab} 5
Red alder ^x	66.2 (25.8) ^a 22	48.2 (14.0) ^b 38	53.7 (17.5) ^{ab} 26	-
Cherry	- 0	34.0 (2.8) 2	69.0 (-) 1	-
Dogwood	31.0 (7.0) 3	33.5 (0.7) 2	-	59.0 (-) 1
Douglas-fir ^x	68.9 (36.0) ^a 14	42.9 (22.9) ^b 15	62.4 (36.1) ^{ab} 21	-
Western hemlock	57.6 (18.7) 19	39.0 (9.5) 3	57.9 (39.8) 17	65.6 (30.5) 8
Western red cedar	-	39.5 (14.0) 6	42.9 (23.7) 25	59.5 (15.2) 6
Grand fir	38.4 (14.3) 14	37.9 (11.0) 7	41.4 (6.8) 5	-
All trees ^x	51.5 (28.9) ^a 151	38.8 (17.8) ^b 174	43.7 (25.4) ^b 203	62.0 (21.3) ^a 21
Conifers ^x	52.1 (27.7) ^a 52	36.7 (19.0) ^b 46	46.1 (30.0) ^{ab} 99	62.1 (23.8) ^a 16
Deciduous trees	51.1 (29.6) ^a 99	39.5 (17.4) ^b 128	41.4 (20.1) ^b 104	61.7 (15.1) ^{ab} 6
Shade tolerant ^x	51.3 (23.7) ^{ab} 22	37.1 (19.2) ^a 14	42.2 (28.1) ^a 64	62.1 (23.8) ^b 15
Shade intolerant	57.9 (29.6) ^a 100	41.0 (17.6) ^b 134	47.9 (23.1) ^b 112	62.2 (16.8) ^{ab} 5

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses.

Table 6-7. Mean number of dead trees (all results from 1996) of different species (standard deviation) for the different sections. Tree species that were present in less than two sections were considered rare and omitted.

	City	Mid	Far	OG
Bigleaf maple*	4.4 (3.3) ^{ab}	4.5 (4.6) ^{ab}	7.3 (5.4) ^a	0.7 (0.6) ^b
Red alder ^x	0.4 (0.8) ^b	2.0 (2.1) ^a	0.8(1.0) ^b	0.0 (0.1) ^b
Cherry	0.1 (0.3)	1.0 (2.0)	1.0 (2.0)	-
Cascara	0.5 (1.3)	0.2 (0.4)	0.3 (0.5)	-
Dogwood	0.1 (0.4)	0.0 (0.2)	0.0 (0.2)	0.3 (0.6)
Douglas-fir	0.4 (0.8)	0.6 (1.2)	1.3 (2.1)	2.3 (1.5)
Western hemlock ^x	0.3 (0.9) ^a	0.3 (1.2) ^a	1.6 (2.2) ^b	3.7 (1.5) ^b
Western red cedar ^x	0.0 (-) ^a	0.0 (0.2) ^a	1.4 (1.8) ^b	2.7 (3.1) ^b
Grand fir	0.3 (1.0)	0.0 (0.2)	0.4 (1.3)	-
# Dead trees of all species ^x	6.8 (3.6) ^a	9.1 (6.5) ^a	14.5 (7.9) ^b	9.7 (2.3) ^{ab}
# Dead conifers ^x	1.0 (1.3) ^a	1.1 (2.1) ^a	4.7 (4.7) ^b	8.7 (2.3) ^b
# Dead deciduous trees ^x	5.5 (3.1) ^{ab}	7.8 (6.1) ^{ab}	9.5 (5.7) ^a	1.0 (0.0) ^b
# Dead shade tolerant ^x	0.3 (0.9) ^a	0.3 (1.3) ^a	3.1 (3.0) ^b	6.3 (1.5) ^b
# Dead shade intolerant	5.2 (3.1)	7.1 (5.6)	9.4 (5.9)	3.0 (2.0)
Mortality rate 1993-1996	7.1 (6.2)	6.2 (5.5)	8.5 (6.3)	6.9 (5.7)
% of dead trees in 1996	24 (10) ^a	25 (11) ^{ab}	34 (13) ^b	24 (7) ^{ab}
# dying 1993-1996	1.9 (1.6)	2.3 (2.0)	3.4 (2.1)	3.0 (2.6)
n	24	24	24	3

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses; a superscript of "*" denotes that the natural log of the variable + 1 was used.

Figure 6-2. Mean increment growth rate over the last seven decades for Douglas-fir and bigleaf maple in the different sections of the study

over the years in the growth rates. For results from ANOVA and Bonnferoni/Dunn analysis, see Appendix G.

The live crown ratio (LCR) for all trees was significantly greater in the old-growth and city sections than in the far section (Table 6-8). Bigleaf maples had a greater live crown ratio in the city and middle sections, and Douglas-firs, western red cedars, and grand firs were highest in the middle section. When the live crown ratio of only upper canopy trees was examined, it was greatest for bigleaf maples in the city section and for Douglas-firs in the middle and far sections.

DISCUSSION

Many of the variables related to tree demographics and vigor were correlated with one or more of the urbanization variables. My results showed a dearth of recent tree regeneration as measured by the number of seedlings at my sites. Most sites did not have any seedlings, and those sites that did have seedlings had a preponderance of early seral species. Only one site (site 7 in the far section) had any seedlings (four) of a climax species -- western red cedar. The distribution of seedlings was not correlated with any measure of urbanization. The low numbers of seedlings could have been a function of weather or other environmental factors. It could also have been an artifact of my sampling procedure, although I did not observe many seedlings at any location in the park.

Although seedlings were scarce, I found saplings at most sites indicating that some regeneration of trees has occurred. The density and composition of saplings showed correlations with urbanization. The densities of larger saplings of all tree species,

Table 6-8. Live crown ratio (LCR) for trees of different species (standard deviation) for the different sections. Tree species that were present in less than two sections or for which the total n was less than 12 were considered infrequent or rare species and were omitted. n is reported for each variable.

Species	City	Mid	Far	OG
Bigleaf maple	0.42 (0.30) ^a 318	0.42 (0.30) ^a 364	0.29 (0.26) ^b 416	0.49 (0.28) ^{ab} 11
Red alder	0.32 (0.16) 61	0.31 (0.23) 164	0.31 (0.18) 91	-
Cherry	0.00 (-) 3	0.10 (0.17) 28	0.16 (0.26) 29	-
Dogwood	0.43 (0.19) 10	0.49 (0.16) 8	0.13 (0.18) 2	0.30 (0.19) 5
Douglas-fir	0.50 (0.24) ^{ab} 64	0.55 (0.25) ^a 99	0.43 (0.28) ^b 108	0.36 (0.27) ^b 25
Western hemlock	0.60 (0.28) 54	0.74 (0.13) 6	0.52 (0.35) 113	0.45 (0.29) 46
Western red cedar	-	0.84 (0.18) ^a 35	0.63 (0.32) ^b 155	0.57 (0.26) ^b 38
Grand fir	0.41 (0.19) ^a 60	0.57 (0.18) ^b 16	0.28 (0.23) ^c 32	-
All trees	0.44 (0.28) ^a 633	0.43 (0.30) ^{ab} 768	0.40 (0.31) ^b 1012	0.47 (0.29) ^a 126
Upper canopy maple	0.58 (0.14) ^a 45	0.54 (0.23) ^a 80	0.46 (0.12) ^b 79	-
Upper canopy Douglas- fir	0.59 (0.13) ^a 43	0.69 (0.15) ^b 40	0.64 (0.09) ^{ab} 46	0.50 (0.16) ^c 18
Upper canopy alder	0.33 (0.12) 22	0.38 (0.14) 45	0.35 (0.11) 45	-
Upper canopy all species	0.54 (0.17) 114	0.54 (0.22) 171	0.52 (0.18) 205	0.54 (0.18) 22
Upper canopy shade intolerant	0.53 (0.16) 110	0.53 (0.22) 165	0.48 (0.15) 170	0.50 (0.16) 18

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

and of both sizes of red cedar saplings, coniferous saplings, and shade-tolerant saplings were lower at more urban sites as indicated by the distance from downtown Portland. The results suggest that urbanization may suppress the recruitment of trees, especially shade tolerant species, in Forest Park. This assertion is supported by the results of multiple regression that revealed that the densities of large saplings, small cedar saplings, small coniferous saplings, and shade tolerant saplings were all associated with edge habitat as indicated by correlations with the urbanization variables. An increase in density of saplings and small trees is a documented edge effect (Chen et al 1992; Chen et al 1995; Matlack 1993a). Since the city section of the park had more edge (Figure 5-2, page 75), one could reasonably expect more saplings in the city section of Forest Park due to the edge effect, but, the opposite was found. The decreased density of saplings at more urban sites could be the result of a lack of seed source or higher rates of seed or seedling predation in the city section of the park, or it may be due to the effects of pollution or human foot traffic that would be greater in more urban areas. The consequences of such human disturbances appear to be overriding any potential edge effects that may be resulting from fragmentation. The apparent reduction in tree regeneration is in agreement with the findings of Rudnický and McDonnell (1989) and Airola and Buchholz (1984), although I did not find a concomitant increase in the number of saplings of "weedy" tree species. This lower density of saplings in the city section, especially of shade-tolerant species, should be investigated further.

Statistical examination of the species composition in each age class also revealed correlations with urbanization. I found decreased densities of several sizes of trees including smaller (10-20 cm dbh) western red cedars and both smaller shade-tolerant

and shade-intolerant trees at more urban sites. Variables associated with edge habitat in this study included 10-20 cm sized western red cedars and shade-tolerant trees, and 20-40 cm sized red alders. I found the same reduction in the density of smaller sized trees in the city section that I noted for saplings. The decreased density of smaller trees in the city section may be the result of long term suppression of regeneration as suggested by the results for saplings. If this is the case, it means that some aspect or aspects of urbanization may have been suppressing tree recruitment for a longer period of time.

The negative correlation between the distance from downtown Portland and the density of small trees suggests that small trees are more affected by urbanization than are larger (older) trees. This result also suggests that some aspect of urbanization other than habitat fragmentation is important in the distribution of small trees. The fact that I found more small dead trees in the far section than the city suggests that increased mortality is not the reason for having fewer young trees in the city section. However, the lower density of saplings and smaller trees at sites near the city may indicate that urbanization is suppressing tree recruitment in the more urban parts of Forest Park. The causes for this finding should be investigated further in another study.

Distribution of the trees into various size classes revealed that the old-growth stand exhibited a more or less reverse-J shaped curve, whereas the trees in the other areas of Forest Park (city section is shown, but middle and far section graphs appear similar) exhibited a uni- or bi-modal shaped curve (Figure 6-3). In addition, the old-

Figure 6-3. Size class distribution of trees in the city section of Forest Park and in the old-growth stand.

growth stand is dominated by coniferous trees, whereas Forest Park is dominated by deciduous trees.

Closer examination of the distribution of tree species in the various size classes also revealed that different seral stages were present at my sites, although all were in a closed canopy stage (Franklin and Dyrness 1988; Houle 1996). To examine this idea more closely, I composed five different successional stages (seres), ranging from sites dominated by shade-intolerant species to mature forest and spanning the successional gradient present at my sites. The earliest seral stage was represented by sites having canopies dominated by shade-intolerant species such as bigleaf maple, Douglas-fir, and/or red alder with few to no shade-tolerant understory trees. The second seral stage was also dominated by bigleaf maple, Douglas-fir, and/or red alder, but small shade-tolerant trees were relatively common in the understory. The third seral stage was represented by sites that still had many bigleaf maple, Douglas-fir, and/or red alder in the canopy, but which also had sizable shade-tolerant trees in the understory and at the mid-canopy level of the forest. The second and third stages show understory re-initiation. The fourth stage was dominated by large bigleaf maple, western hemlock, and western red cedar, the last two being shade-tolerant species of trees. The fifth seral stage -- mature forest -- was dominated by very large, old Douglas-firs and large western hemlocks, with a multilayered understory of variously sized western hemlocks and western red cedars. This stage was found only at the old-growth stand. I classified each site into one of the five seres on the basis of the species distribution and relative age structure found at each location (Table 6-9). Sites in the city section had an average seral stage of 1.8; sites in the

Table 6-9. Distribution of sites into different seral stage categories as determined in this study.

<u>Seral Stage</u>	<u>Site Numbers in the Following Sections</u>			
	City	Mid	Far	OG
1	3, 5, 12, 14	11, 32, 37	41	-
2	1, 13	23	2, 6, 7, 15, 19, 39	-
3	28, 30	17, 18, 24, 36	-	-
4	-	-	22	-
5	-	-	-	og

middle section had 2.0; sites in the far section averaged 2.1; and the old-growth stand was 5.0 (Table 6-10), showing, that in general, the successional status of sites increased with distance from Portland, although these were not significant differences.

The age structures of trees at each site revealed that the majority of sites were in the first two (earlier) stages of succession (Tables 6-9 and 6-10), as is much of Forest Park (Portland Parks and Recreation 1995). These two stages are characterized by having minimal understory development and, at least in Forest Park, most likely result from a history of punctuated disturbances such as logging, wood cutting, and/or fires. Logging and woodcutting are obviously human disturbances. In Forest Park, recent major fires were caused by human activity. For example, the last major fire in 1951 was caused by an escaped campfire (Robinson 1978). Major disturbances such as fire and logging produced the mosaic-like pattern of successional stages seen in the forest, but they are not the whole story. The dominance of much of the park by red alder and bigleaf maple has probably been caused by the history of continual disturbance of the land. Normally, after a disturbance in the Pacific Northwest, Douglas-fir would be the major early seral tree species to invade an upland site. However, severe and/or continual disturbance leads to the depletion of soil nutrition (especially nitrogen), and frequently results in hardwoods replacing softwood coniferous species (Franklin and Pechanec 1967; Houle 1996; Newton et al. 1967). The complete dominance of hardwoods in the park has precluded the invasion of most sites by Douglas-fir, a shade-intolerant species. Instead, if a seed source is available, shade-tolerant species eventually begin to invade, as is seen at many sites with young understory trees. As the hardwoods

Table 6-10. Total number of sites in each seral stage in each section as well as the mean seral stage (standard deviation) found in each section.

Seral Stage	Total Number of Sites in the Following Sections			
	City	Mid	Far	OG
1	4	3	1	0
2	2	1	6	0
3	2	4	0	0
4	0	0	1	0
5	0	0	0	1
Mean seral stage	1.8 (0.9) ^a	2.0 (0.9) ^a	2.1 (0.8) ^a	5.0 (-) ^b
n	8	8	8	1

Results of Bonnferoni/Dunn testing is shown for mean seral stage as superscripts for means that are significantly different from one another. Means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b).

age and begin to fall due to disease and old age, the corresponding openings in the canopy release those understory trees, allowing them to overtop the remaining hardwoods and eventually become dominant (Franklin and Pechanec 1967; Houle 1996; Newton et al. 1967). I expect that Forest Park, if undisturbed, will circumvent much of the Douglas-fir stage, and instead succeed directly to a climax forest dominated by shade-tolerant species such as western hemlock and western red cedar. These two trees are present in the understory at many sites. However, sites that are dominated by extensive stands of pure hardwoods may lack appropriate sources of shade-tolerant seeds, and may need to have young trees planted since the Forest Park master plan advocates for the eventual successional culmination to a mature or old-growth coniferous forest (Portland Parks and Recreation 1995).

To determine the date of the last major disturbance at each site (i.e., fire or logging), I assumed that the date of such a disturbance corresponded to the average age of shade-intolerant trees on the site. The old-growth stand showed evidence of fire (a major disturbance). Assuming that the bigleaf maples at this site began their growth after that event and since the mean age of the bigleaf maples was 62 years, I estimated that a fire had swept through the old-growth in the early 1930s even though the largest Douglas-firs at the site were estimated to be approximately 425 years old.

My data revealed that study sites in Forest Park were younger than the old-growth stand with no site having any trees approaching the size and/or estimated age of the large Douglas-firs in the old-growth stand. Most sites in Forest Park appeared to have experienced the last major disturbance (logging and/or fire) between 38 and 57 years ago. Sites in the city section experienced their last major disturbance on

average about 52 years ago. The tree ages support the hypothesis that the city section is in an older successional stage than is the far section of the park since the far section averaged only 44 years since the last major disturbance. Trees at sites in the city section were significantly older than trees at sites in either the middle or far sections of Forest Park. A map at Portland Parks and Recreation showed that the far section had more areas of recent logging and/or fire than the city section. The average age of trees at sites in the middle section (39 years) coincided with the date of the big fire of 1951, although sites 18 and 32 lay outside of the boundary of that fire and had older trees. Another piece of evidence regarding the disturbance history of the middle section of the park was imparted by bigleaf maple. Bigleaf maple frequently responds to fire or cutting by suckering (sending up shoots from the existing roots). Often, more than one shoot sprouts from the existing root, frequently leading to trees with multiple trunks. ANOVA demonstrates there are significantly more bigleaf maples with multiple trunks in the middle section (Figure 6-4). This, along with a greater abundance of red alders, suggests that the middle section has experienced a major disturbance.

Most likely, the successional status at each site (Table 6-9) is related to the past land use history of Forest Park. However, when the successional status at each site is compared to the relative ages of the dominant trees in each section, the city section appears to be at an earlier successional stage than would be expected based on the length of time since the last major disturbance. If suppression of shade-tolerant saplings had occurred at more urban sites, sites in the city section would have less understory development. The lack of understory development would make it difficult to ascertain the relative time since the last major disturbance for sites in that

Figure 6-4. The mean number of multi-trunk bigleaf maples in the different sections of Forest Park and the old-growth stand.

section of the park from stand structure alone. In a closed canopy forest, most seedlings and saplings should be shade-tolerant species. If suppression of small trees occurred for several decades, the population age curve could become skewed. A reduction in the establishment of shade-tolerant trees in urban areas of the forest would disrupt normal successional processes of stand re-initiation at more urban sites. Such a mechanism would explain how the city section could be older than the far section of Forest Park, even though there was a paucity of shade-tolerant trees in more urban areas. The scarcity of shade-tolerant trees in the forest understory poses questions as to the future of much of Forest Park as the shade-intolerant species reach the end of their normal life spans and die. Many of the red alders in the park are approaching that age now and many have rotten cores that may contribute to their demise in future wind and ice storms. The death of such trees opens the canopy and normally allows release of shade-tolerant understory trees. Sites without an understory of shade-tolerant species may remain dominated by hardwood species, or perhaps they will shift to a Douglas-fir forest if the appropriate seed source is available.

The mortality of trees is another major component of succession in the forest. Tree mortality was greater at sites further from Portland and increased with distance from the city for western hemlock, western red cedar, conifers, and shade-tolerant species. In addition, increased numbers of dead western hemlocks and dead conifers were associated with forest interior habitat. The likeliest explanation for this increased mortality is self-thinning or stem exclusion. As overstory trees become more dominant during succession and eventually reach the closed canopy stage, their canopies become denser, leading to the suppression and frequently the mortality of

understory trees, especially those of shade-intolerant species (Franklin and Dyrness 1988). This argument is supported by the fact that most dead trees in Forest Park are shade-intolerant species. Only in the old-growth stand did I find more dead trees of shade-tolerant species. The death of trees in 1996 is positively correlated to the density of trees in that same year by a second order polynomial regression (Figure 6-5). The normal successional processes of self-thinning appear to be overriding any effects of urbanization as a factor in tree mortality in the park. As self-thinning occurs understory trees are released from suppression and can grow to overtop the present canopy of shade-intolerant tree species such as red alder and bigleaf maple. The far section, with more shade-tolerant saplings in the understory, appears to be on a normal successional pathway.

The only measures of tree vigor (examined as increment growth rate) that were associated with urbanization (as measured by the distance from downtown Portland) were the increment growth rates of western red cedars 10-20 years ago and western hemlocks 20 to 30 years ago. The increment growth rate for western red cedar and western hemlock during these time periods was lower at sites nearer the city. The finding of depressed growth in western red cedar 10-20 years ago and western hemlock 20 to 30 years ago at more urban sites may be an indication of the effects of urbanization upon tree growth and health. However, more data are needed to substantiate this finding before such conclusions can be reached since depressed growth rate could also be the result of earlier canopy closure in the "older" city section of the park.

Figure 6-5. Graph of tree mortality (number of dead trees in 1996) versus the total number of trees at sites in 1996. The second order polynomial regression formula was $y = 1.28x + 0.10x + 0.004x^2$; $R^2 = 0.75$; $p < 0.0001$.

The growth rate time series (Figure 6-2, page 96) was indicative of the fact that increment growth rates had been greatest about 40 to 50 years ago, probably due to the release of trees after fire and/or logging. For all sections of Forest Park, the rate has declined each subsequent decade as the canopy has closed. Only the old-growth stand remained more or less consistent (and low) over the years. Since large Douglas-fir trees are less susceptible to fire than are many other species, and since these trees dominate the canopy in the old-growth stand, it is likely that the old growth did not experience substantial canopy opening after the fire.

The other indicator of vigor that I calculated was the live crown ratio. The live crown for upper canopy trees was reduced at sites nearer to Portland. This may be another indication that trees at sites closer to downtown Portland have lower vigor than trees at more rural sites. The decreased live crown ratio may be due to canopy closure at the "older", more urban sites as a normal consequence of succession. However, the reduced live crown ratios and the reduced increment growth rates at more urban sites may also be the result of some factor of urbanization. Pollution, especially by heavy metals associated with automobile exhaust, has been demonstrated to result in reductions of tree health (Airola and Buchholz 1984; Ashby and Fritts 1972; McLaughlin et al. 1983; Nash et al. 1975; Pouyat et al. 1994; Pouyat et al. 1995a, 1995b; Supuka 1994; Thompson 1981; White and McDonnell 1988; Yost et al. 1991). My findings could be an indication of the effects of pollution on tree health in the more urban areas of Portland that could possibly cause reductions in increment growth and live crown canopy. Such effects could also account for the suppression

of tree recruitment in the city section of the park. However, research on this specific topic would be needed to determine whether such an effect exists.

CHAPTER 7

THE EFFECTS OF HUMAN DISTURBANCE AND URBANIZATION UPON
THE RICHNESS, ABUNDANCE, AND SPECIES DIVERSITY OF BIRDS

Extensive research has been published on the effects of human disturbance upon bird communities. Much of the interest, which began in the 1970's, grew out of concern for the seeming decline of certain neotropical migratory bird species throughout the eastern and midwestern regions of the United States (Askins and Philbrick 1987; Galli et al. 1976; Terborgh 1989; Whitcomb et al. 1981; Wilcove 1985). This concern led to research into the reasons for the decline (Aldrich and Coffin 1980; Askins 1994; Askins 1995; Askins and Philbrick 1987; Beissinger and Osborne 1984; Blake et al. 1994; Blake and Karr 1987; Bollinger and Linder 1994; Brown and Robinson 1996; Donovan et al. 1995a, 1995b; Edgar and Kershaw 1994; Friesen et al. 1995; Graber and Graber 1983; Hagan et al. 1996; Herkert 1995; Kroodsma 1984; Lent and Capen 1995; Levenson 1981; Lynch and Whigham 1984; Martin et al. 1996; Mauer and Heywood 1993; McIntyre 1995; Peterjohn and Sauer 1994; Rappole and McDonald 1994; Rich et al. 1994; Robinson and Holmes 1984; Robinson et al. 1995; Stouffer and Bierregaard 1995; Taper et al. 1995; Temple 1986; Temple and Wilcox 1986; Terborgh 1980; Terborgh 1989; Terborgh 1992; Verner 1986; Wilcove 1985; Yahner and Mahan 1996). Much of the research has centered on the effects of habitat fragmentation on bird species richness, abundance, and diversity. The research has focused attention on several factors that are believed to have played a role in the decline of migratory songbirds.

Many scientists have sought to determine whether the theory of island biogeography explains observed reductions in avian species diversity in forest remnants. According to this theory, larger islands will support more species at equilibrium than similarly situated smaller islands, and islands closer to the mainland should support more species than similarly sized islands further from the mainland (MacArthur 1972; MacArthur and Wilson 1967). Even though forest remnants are often referred to as "islands", the theory of island biogeography has proved inadequate to explain many of the observed changes. In part, this is believed to be because the matrix separating the "islands" is, unlike water, capable of supporting terrestrial life. Most researchers have found greater diversity in larger forests (Askins 1994, 1995; Blake 1983; Blake and Karr 1987; DeGraaf 1991, 1995; Donovan et al. 1995b; Gavareski 1976; Hoover et al. 1995; Lent and Capen 1995; Lynch and Whigham 1984; McIntyre 1995; McLellan et al. 1986; Robinson et al. 1995; Rosenberg and Raphael 1986; Temple 1986; Tilghman 1987; Verner 1986; Wiens 1994; Wilcove 1985), although neither area (size) nor distance from the nearest large forest ("the mainland") have been shown adequately to predict the number of species a forest of a given size can support (Blake 1983; Blake and Karr 1987; Blondel 1985; Davis and Glick 1978; Dzwonko and Loster 1988, 1989; Gavareski 1979; Haila 1986; Levenson 1981; Lovejoy and Oren 1981; McLellan et al. 1986; Rowntree 1986; Tilghman 1987; Tramer and Subrweir 1975; Weaver and Kellman 1981; Whitcomb et al. 1981; Zacharias and Brandes 1990). There does, however, seem to be a minimum critical size that a forest remnant must be in order to support a near normal avian biodiversity (Blake 1983; Dzwonko and Loster 1989; Gotfryd and Hansell 1986; Johnson et al. 1981; Ranney et al. 1981; Rosenberg and Raphael 1986; Temple 1986; Tilghman

1987; Whitcomb et al. 1981). For example, a Douglas-fir forest of approximately square or circular shape should be a minimum of 64 hectares (Chen et al. 1995; Franklin and Forman 1987). But as described earlier, the minimum area is as much a function of shape as it is of size. Forests with small area to edge ratios function as edge communities (Blake 1983; Gotfryd and Hansell 1986; Harris 1984; Johnson et al. 1981; Ranney et al. 1981; Rosenberg and Raphael 1986; Temple 1986; Tilghman 1987; Whitcomb et al. 1981). As forest land continues to be cleared, the effects of habitat fragmentation and the increasing amounts of edge habitat are believed to affect adversely many species of birds through a variety of mechanisms.

One consequence of forest fragmentation is increased predation on nests, particularly those of ground nesting birds. Many predators, such as raccoons, opossums, crows, and jays, are edge associated species. Most of these species are present in higher densities in smaller forest fragments and have been shown to significantly decrease ground nesting success (Angelstam and Andren 1988; Askins 1994; Askins 1995; Blake 1983; Bollinger and Linder 1994; Brown and Robinson 1996; Craig 1997; DeGraaf 1995; Diehl 1986; Donovan et al. 1995a; Hagan et al. 1996; Hoover et al. 1995; Lehmkuhl and Ruggiero 1991; Leimgruber et al. 1994; Linder and Bollinger 1995; Martin and Clobert 1996; Martin et al. 1996; McLellan et al. 1986; Rich et al. 1994; Robinson et al. 1995; Rosenberg et al. 1987; Terborgh 1989; Whitcomb et al. 1981; Wiens 1994; Wilcove 1985; Yahner and Mahan 1996). Wilcove (1985) demonstrated experimentally that in small forest fragments (less than 10 hectares), predation rates on ground nesting birds could reach 100 percent.

Another major consequence of forest fragmentation is increased rates of brood parasitism by brown-headed cowbirds. Brown-headed cowbirds lay their eggs in the nests of other bird species to the detriment of the host species' young. Although the range of the brown-headed cowbird was once limited to the Great Plains, it has expanded dramatically throughout the last century to its present range covering most of North America, an expansion that followed the advancement of European settlement. It has been reported that brown-headed cowbirds have successfully parasitized 144 species of birds and have been observed to lay eggs in over 200 different species' nests (Ehrlich et al. 1988; Robbins et al. 1983; Terborgh 1989). Female brown-headed cowbirds lay their eggs in the nests of forest species, but forage in open areas such as roadside strips, yards, pastures, and grassy corridors and so need access to forest edge. As forest area decreases, the ability of brown-headed cowbirds to invade and parasitize nests increases. They are nearly ubiquitous in forested areas of all sizes except for very large expanses of contiguous forest (Askins 1995; Brittingham and Temple 1983; Coker and Capen 1995; Terborgh 1989; Wilcove 1985). Research in many areas of the country has demonstrated that brown-headed cowbirds are more numerous in smaller fragments where they significantly reduce the breeding success of native birds (Askins 1995; Bollinger and Linder 1994; Brown and Robinson 1996; Coker and Capen 1995; Ehrlich et al. 1988; Gustafson and Crow 1994; Hoover et al. 1995; Linder and Bollinger 1995; Rich et al. 1994; Robinson et al. 1995; Terborgh 1989; Wilcove 1985). They have already been implicated in the near-extinction of bird species such as Kirtland's warbler (Mayfield 1977) and the least Bell's vireo (Goldwasser et al. 1980), and are expected to impact seriously many other woodland species (Askins 1995; Bollinger and Linder 1994; Brown and Robinson 1996; Coker and Capen 1995; Ehrlich et al.

1988; Gustafson and Crow 1994; Hoover et al. 1995; Linder and Bollinger 1995; Rich et al. 1994; Robinson et al. 1995; Terborgh 1989; Wilcove 1985).

Not only has habitat fragmentation of forests in North America been blamed for bird species reductions, but loss and fragmentation of habitat in neotropical regions has also been implicated in the decline of certain bird species. As neotropical land is cleared of native vegetation, neotropical migratory species of birds lose wintering habitat. Loss of wintering habitat may cause reductions in bird species population numbers (Conway et al. 1995; Hutto 1980; Karr and Roth 1971; Mauer and Heywood 1993; Rappole and McDonald 1994; Terborgh 1980; Terborgh 1989; Wilcove 1985; Willson et al. 1994). It is widely believed that loss of winter habitat is the major cause of the probable extinction of Bachman's warbler, a neotropical migrant that over-wintered primarily in forests in Cuba before they were cleared for sugar plantations (Terborgh 1989). Further evidence supporting tropical deforestation as being important is that many neotropical migratory birds are experiencing population declines, whereas populations of many permanent resident species and short-distance migrant species are remaining stable or even increasing (Conway et al. 1995; Karr and Roth 1971; Terborgh 1989; Wilcove 1985; Willson et al. 1994).

Whereas global land use patterns are cited as a probable cause for the decline of many species, increased local disturbances, many of which are amplified by the process of urbanization, are also believed to have an impact upon bird abundance (density), bird species richness (the number of species), and bird species diversity (H'). Bird abundance has been reported to increase with disturbance (Askins and Philbrick

1987; Davis and Glick 1978; DeGraaf 1991; Lehmkuhl et al. 1991; Mills et al. 1989, 1991; Rosenberg et al. 1987). Bird species diversity (BSD) has been found either to increase with disturbance (Aldrich and Coffin 1980; Blake 1983; Huff and Raley 1991; Lehmkuhl et al. 1991), or decrease (Davis and Glick 1978; Edgar and Kershaw 1994; Gavareski 1979; Lancaster and Rees 1979; McIntyre 1995; Mills et al. 1989; Tilghman 1987). The intermediate disturbance hypothesis may explain these findings as the highest diversity is often reached at intermediate levels of disturbance or development (Blair 1996; Lent and Capen 1995). In a similar fashion, bird species richness has been reported either to increase (Gotfryd and Hansell 1986; Lehmkuhl et al. 1991; Tilghman, 1987) or decrease (Beissinger and Osborne 1984; Blake 1983; Davis and Glick 1978; DeGraaf 1991; Fugate 1994; Johnson et al. 1981; Levenson 1981; Rowntree 1986; Stouffer and Bierregaard 1995; Verner 1986; Wentworth and Telfer 1993) with disturbance. The changes in abundance, BSD, and richness are often attributed to an influx of "edge" and non-native species (Aldrich and Coffin 1980; Askins and Philbrick 1987; Beissinger and Osborne 1984; Blair 1996; Davis and Glick 1978; DeGraaf and Wentworth 1986; Gavareski 1976; Harris 1984; Kroodsma 1984; Lancaster and Rees 1979; Lehmkuhl and Ruggiero 1991; Lent and Capen 1995; McIntyre 1995; McLellan et al. 1986; Mills et al. 1989, 1991; Rich et al. 1994; Rosenberg and Raphael 1986; Rosenberg et al. 1987; Terborgh 1989; Whitcomb et al. 1981; Wilcove 1985), often with a concomitant loss of forest species (Aldrich and Coffin 1980; Askins 1995; Askins and Philbrick 1987; Blair 1996; Blake et al. 1994; Bollinger and Linder 1994; Donovan et al. 1995a; Hoover et al. 1995; Keller and Anderson 1992; Kroodsma 1984; Lehmkuhl and Ruggiero 1991; Lent and Capen 1995; McIntyre 1995; McLellan et al. 1986;

Rosenberg and Raphael 1986; Stouffer and Bierregaard 1995; Temple and Wilcox 1986; Terborgh 1989; Wilcove 1985).

Analysis of avian community structure across urban land use gradients -- usually industrial and commercial property, residential areas of various densities and ages, and/or open spaces such as golf courses or parks -- has revealed shifts in bird abundance, richness, and species diversity. Most researchers have reported increased bird density and decreased bird species diversity with increasing urbanization (Beissinger and Osborne 1984; Blair 1996; Davis and Glick 1978; DeGraaf 1991; DeGraaf and Wentworth 1986; Edgar and Kershaw 1994; Gavareski 1976; Kubantsev and Kolyakin 1995; Lancaster and Rees 1979; Mills et al. 1989, 1991; Reijnen et al. 1995; Rosenberg et al. 1987). Most frequently, there is a loss of native, forest interior species (e.g., western wood-peewee and Hutton's vireo (Blair 1996); black and white warbler and ovenbirds (DeGraaf and Wentworth 1986)), accompanied by a gain in non-native, omnivorous species such as house sparrows, European starlings, and rock doves. As urbanization increases, the abundance of a single species may reach 100 percent (Askins and Philbrick 1987; Beissinger and Osborne 1984; DeGraaf 1991; Edgar and Kershaw 1994; Gavareski 1976; Lancaster and Rees 1979; Mills et al. 1989; Rosenberg et al. 1987). Blair (1996) recently compared bird species richness, density, and diversity along an urbanization gradient from relatively pristine forest to highly urban habitat. He found that moderate development increased avian species diversity by increasing the number of "widely distributed species" such as mourning doves and house finches at the expense of native habitat specialists. As urbanization increased, both total and native specialist species diversity decreased (Blair 1996). Friesen et al. (1995) compared the avian

communities in woodlots of differing sizes that were surrounded by various degrees of urbanization. They found that regardless of the size of the woodlot, diversity and abundance of neotropical migratory bird species was negatively correlated with the number of surrounding houses. The authors speculated that forest interior birds were lost, not only because of the effects of fragmentation, but due to other unidentifiable "negative effects" associated with nearby urban habitat (Friesen et al. 1995).

Many of the previous studies examined changes in the avian community in terms of guild structure. A guild is a functional group of species that have similar habitat and resource use. Most often, bird species are placed into guilds on the basis of the food eaten, the portion of the ecosystem where the food is gathered, and/or food gathering behavior (DeGraaf and Wentworth 1986; Ehrlich et al. 1988). Although a bird's guild often changes with the seasons, the guild occupied during the breeding season is the one usually examined (Abrams 1991; Beissinger and Osborne 1984; DeGraaf 1991; DeGraaf et al. 1991; DeGraaf and Wentworth 1986; Huff et al. 1991; Huff and Raley 1991; Lewke 1982; Manuwal 1991; Manuwal and Huff 1987; Willson 1970, 1974). Bird guilds in urban or disturbed areas become increasingly dominated by omnivores and ground-gleaning species at the expense of more specialized foliage, bark, and/or aerial gleaners (Aldrich and Coffin 1980; Beissinger and Osborne 1984; Blake 1983; Block et al. 1986; Carey et al. 1991; Davis and Glick 1978; DeGraaf 1991; DeGraaf and Wentworth 1986; Lancaster and Rees 1979; Lent and Capen 1995; Manuwal 1991; Manuwal and Huff 1987; Rosenberg et al. 1987).

Avian communities are also commonly examined in terms of the migratory status of bird species. This has become increasingly important because of evidence that tropical deforestation may have detrimental effects on neotropical migratory bird species diversity and abundance (Conway et al. 1995; Hutto 1980; Karr and Roth 1971; Mauer and Heywood 1993; Rappole and McDonald 1994; Terborgh 1980; Terborgh 1989; Wilcove 1985; Willson et al. 1994). The observed decline in neotropical migratory species diversity and abundance has been documented in the eastern and central portions of the United States. In these areas, neotropical species account for 70 to 75 percent of the breeding bird population. In the western United States, however, the percent of neotropical bird species is lower. In old-growth Douglas-fir forests, 60 to 100 percent of breeding season birds are members of resident bird species (Carey et al. 1991; Harris 1984; Huff and Raley 1991; Manuwal 1991; Manuwal and Huff 1987; Morse 1980). In addition, the neotropical species that summer in the western United States coniferous forests commonly winter in Mexico rather than Central or South America where disturbances have been greater (Hutto 1980; Manuwal 1991; Terborgh 1980). However, as all tropical areas become increasingly altered by human disturbance, there are potential implications for reductions in neotropical bird abundance and diversity in the Pacific Northwest.

Many researchers have also attempted to correlate bird species richness, abundance, and diversity with plant community structure. For example, BSD has been correlated with foliage height diversity (FHD) (Bersier and Meyer 1994; Karr and Roth 1971; Lancaster and Rees 1979; MacArthur and MacArthur 1961; Mills et al 1989, 1991; Willson 1974), plant species richness and diversity (Dobkin and Wilcox 1986), and canopy cover or foliage volume (Beissinger and Osborne 1984; Ito and Fujimaki

1990; O'Brien 1990; Rowntree 1986; Tilghman 1987; Willson 1974). Some scientists have attempted to correlate bird density or abundance with total vegetation volume (TVV) (Mills et al 1989, 1991; Willson 1974), the percent of coniferous or deciduous forest (Carey et al 1991; Clark et al 1983; Goldstein et al 1986; James and Wamer 1982; MacArthur and MacArthur 1961; Tilghman 1987), tree height (Aldrich and Coffin 1980; Mills et al 1989, 1991; O'Brien 1990; Tilghman 1987; Wentworth and Telfer 1993), canopy height or the amount of foliage cover (Clark et al 1983; Sedgwick 1987), and the amount of non-native vegetation (Beissinger and Osborne 1984; DeGraaf 1991; DeGraaf and Wentworth 1986; Gravareski 1976; Mills et al 1989, 1991). Results from these studies have been inconclusive. In this study, such comparison may be important to help elucidate whether observed changes in avian distribution are due to community succession (and the changing plant characteristics that accompany that succession) or if urbanization is playing a role.

METHODS

Bird species richness and abundance were measured using the fixed-radius point method (Manuwal and Carey 1991). Although many biologists prefer Emlen's spot count method (Emlen 1977), scientists in Pacific Northwest forests have found it difficult to walk along a transect over the rough terrain and through thick vegetation while accurately detecting birds. Spot counts and line transect methods have been reported to undercount birds by as much as 40 percent (Manuwal and Carey 1991). In addition, since each of my sites covered a limited area, it was impossible to use the transect method without leaving the site.

For each count, all birds were counted that were seen and/or heard within a 10 minute time period and that were estimated to be within a 50 meter radius extending from the center of the site (Hutto et al. 1986; Manuwal and Carey 1991; Verner 1985). These measurements were taken after waiting silently and unmoving for 5 minutes to ensure that birds were accustomed to the observer's presence. Bird counts were not done during periods of moderate to heavy rain or during episodes of strong winds or heavy fog. Assumptions of the procedure were that all birds were accurately identified, all birds in the circle were counted, the limits of the circle were recognizable, all bird species were equally detectable, and all recorded birds were counted only once per census (Manuwal and Carey 1991). All bird counts were conducted by the same observer.

Birds were counted during the breeding season (May to June) and in the winter (December to February). All breeding season bird counts were conducted between sunrise and 11:00 a.m. During the spring, each site was visited a total of three times each year, with counts at each individual site being conducted at least one week apart. The counts were repeated for three consecutive seasons (1993 - 1995). During spring of 1993, bird counts were conducted between May 8 and June 30; during the spring of 1994, bird counts were conducted between May 14 and June 28; and during the spring of 1995, bird counts were conducted between May 30 and June 30.

All winter bird counts were conducted between sunrise and 3 p.m. because birds are active throughout the daylight hours during the winter season. Only one count per site was done during the 1994 and 1995 seasons, but three counts per site were conducted during the 1996 season. The counts at a given site in 1996 were made at

least one week apart. Counts were conducted between January 10 and January 19 in 1994; between January 4 and January 20 in 1995; and between January 3 and February 20 in 1996. The much longer time period for 1996 was due to the triplicate sampling and to the severe winter storms that delayed field work.

Bird abundance, richness, and diversity were examined using the same statistical analyses as previously discussed -- multiple regression using the same seven independent variables of urbanization plus the covariable and ANOVA using section of the park as the independent variable. Counts during the breeding season were statistically analyzed for total birds counted per year since annual fluctuations were not important to this study. Winter counts, however, were statistically analyzed for mean birds per year because I did not have triplicate samples for the first two seasons.

Bird richness and abundance were also examined for certain "indicator" species. Eight bird species were assigned to each habitat type -- urban, edge, or forest (Table 7-1). Bird species that prefer paved and open lawn areas were considered indicative of cities or urban areas. Such bird species are commonly present in urban settings. The eight bird species that were selected to represent this category were rock doves, house sparrows, European starlings, American robins, house finches, American crows, brown headed cowbirds, and scrub jays. Three of the selected species are exotic, and a fourth, the brown-headed cowbird, is not native to western Oregon, although it is native to North America (Gilligan et al. 1994). These species are referred to in this study as "urban birds". Edge habitats are usually characterized by

Table 7-1. Bird species used as indicator species for the three habitat types: urban, edge, and forest interior. Abbreviations used for the birds are given after each species.

<u>HABITAT TYPE</u>	<u>BIRD INDICATOR SPECIES</u>
Urban	rock dove* (RODO), house sparrow* (HOSP), European starling* (EUST), American robin (AMRO), house finch (HOFI), American crow (AMCR), brown-headed cowbird (BHCO), scrub jay (SCJA)
Edge	spotted towhee (SPTO), olive-sided flycatcher (OSFL), Bewick's wren (BEWR), mourning dove (MODO), northern flicker (NOFL), yellow-rumped warbler (YRWA), purple finch (PUFI), and song sparrow (SOSP)
Forest Interior	band-tailed pigeon (BTPI), hairy woodpecker (HAWO), pileated woodpecker (PIWO), Pacific-slope flycatcher (PSFL), western wood-peewee (WWPE), brown creeper (BRCR), Swainson's thrush (SWTH), black-throated gray warbler (BTWA)

* non-native species

thick brushy areas. Bird species associated with such habitats were chosen as "edge" species. The bird species that were selected to represent edge environments were spotted towhees, olive-sided flycatchers, Bewick's wrens, mourning doves, northern flickers, yellow-rumped warblers, purple finches, and song sparrows. These eight species are referred to in this study as "edge birds". Forest interior bird species are those species rarely found outside of forested environments. The species selected as indicative of forest interior habitat were band-tailed pigeons, hairy woodpeckers, pileated woodpeckers, Pacific-slope flycatchers, western woodpeewees, brown creepers, Swainson's thrushes, and black-throated gray warblers. These bird species are referred to as "forest birds". Although each of the 24 species chosen was used to indicate a certain habitat type, many of the other bird species observed during this study have also been associated with urban, edge, and/or forest habitat by other authors (Blair 1996; Carey et al. 1991; Gilbert and Allwine 1991; Huff and Raley 1991; Lundquist and Mariani 1991; Manuwal 1991; Ruggiero et al. 1991; Whitcomb et al. 1981).

I also analyzed bird abundance by guild organization and migratory status. I categorized birds into guilds based on food gathering behavior (Ehrlich et al. 1988). The following guilds were used: foliage gleaners (take food items from vegetation), ground gleaners (pick food items from the surface of the ground), bark gleaners (remove invertebrates from bark; includes birds that excavate and drill into bark), aerial gleaners (take food while in flight), and predatory birds (eat other vertebrates). Aerial gleaners includes the categories of gleaning while hovering (e.g., hummingbirds), short flights to catch prey (e.g., Pacific-slope flycatcher), and capturing prey while in continuous flight (e.g., violet-green swallow). Predatory

birds include birds that hunt using either aerial pursuit (chasing and catching birds in the air or on their perch, such as a Cooper's hawk), or swoop (snatching prey off of the ground using talons, such as a northern pygmy-owl) (Ehrlich et al. 1988). A list of birds and their foraging guilds can be found in Appendix H.

Since tropical deforestation is believed to have a major impact on neotropical bird species richness and abundance, I examined migratory status to see whether or not there were any short-term changes observable in Forest Park's avian community and/or if there were any changes correlated with urbanization variables. The migratory status of a bird indicates whether or not, and how far, that species migrates annually. Bird species are often described as being residents, short distance (local) migrants, or neotropical (long distance) migrants (Askins 1995, Askins and Philbrick 1987, Blake et al. 1994, DeGraaf 1991, Ehrlich et al. 1988; Lynch and Whigham 1984). Residents are birds that remain in the same general location throughout the year. Examples of resident bird species in the Portland area include the black-capped chickadee and dark-eyed junco. Short-distance migrants, or local migrants, are bird species that travel a short distance in terms of either latitude or elevation, but do not enter tropical areas. These bird species are present during the winter months in the Portland area, having come from further north or higher elevations. Examples of local migratory species include the American robin and white-crowned sparrow. Neotropical (long-distance) migratory species are those that annually fly to Mexico, Central America, and/or South America. Members of this group will only rarely be seen in the Portland area during winter. Examples include the Swainson's thrush and Wilson's warbler (Ehrlich et al. 1988; Gilligan et al. 1994; Hutto 1980). The migratory status of birds counted in this study is listed in Appendix H.

I analyzed avian species abundance, richness, and diversity, as well as species abundance, richness, and diversity as categorized by both migratory status and guild membership. I analyzed breeding season birds and winter birds. Avian species that were counted less than 30 times total and/or present in less than two sections of the park were considered rare and were not analyzed. All dependent variables were analyzed using multiple regression and ANOVA as described previously. The common and scientific names of all bird species counted in this study, as well as abbreviations used, are listed in Appendix I.

I also compared avian community variables (dependent variables) with a select group of plant community variables (independent variables) using multiple regression. Plant community variables were selected that had been significantly correlated with urbanization in the previous chapters, and that were indicative of successional stage and/or increased numbers of exotic plant species. I tried to select variables that encompassed all plant layers (e.g., I selected variables that measured different characteristics of trees, shrubs, and herbs), as well as foliage height diversity (FHD). Using a correlation matrix, I reduced the number of useful vegetation variables to eight by selecting variables that had less than 60 percent collinearity and that met the requirements of representing different sections of the plant community. The eight plant community variables that were chosen functioned as the independent variables in multiple regression analyses that compared plant community structure with avian distribution. The eight independent variables were: the number of trees (TREE), the number of bigleaf maples (MAPLE), the number of small (10-20 cm dbh) western red cedar trees (CEDAR), the number of shrubs (SHRUB), the total basal area of

thimbleberry (THIMB), the species diversity for the herb layer (HERB), the percent cover by exotic herbs (EXOTIC), and the foliage height diversity (FHD). To meet the assumptions of normality, some of the independent variables were transformed using the same method described earlier (Table 7-2) (Draper and Smith 1966; Jongman et al. 1995; Sokal and Rohlf 1969; Steel and Torrie 1960; Tabachnick and Fidell 1996). These transformations were used consistently in all data analyses where vegetation parameters were compared to avian variables. Dependent variables (avian) that were not normally distributed were also transformed (Draper and Smith 1966; Jongman et al. 1995; Sokal and Rohlf 1969; Steel and Torrie 1960; Tabachnick and Fidell 1996). The transformations are reported in the tables. Regression coefficients comparing urbanization variables with breeding and winter bird variables are reported in Appendix J; regression coefficients comparing avian and vegetative variables are reported in Appendix K.

RESULTS

Breeding Birds (Spring Counts)

Fifty-six species of birds were observed at my sites during the breeding season over the three year period of data collection (Table 7-3). During that time, 6062 individual birds were counted. I counted 1948 birds in the city section, 1915 in the middle section, 1866 in the far section, and 333 in the old-growth.

Several variables relating to breeding season birds were correlated with one or more of the urbanization variables by multiple regression (Table 7-4). The abundances of

Table 7-2. Independent variables of vegetation parameters, their abbreviations as utilized in the paper, and the transformations done to meet assumptions of normality. These transformations were used in all statistical analyses pertaining the vegetation variables.

<u>Vegetation Variable (Abbreviation)</u>	<u>Transformation</u>
Density of trees (TREE)	square root (TREE)
Density of bigleaf maples (MAPLE)	none
Density of w. red cedar 10-20 cm dbh (CEDAR)	square root (CEDAR)
Density of shrubs (SHRUB)	ln (SHRUB + 1)
Total basal area thimbleberry (THIMB)	square root (THIMB)
Herbaceous species diversity (HERB)	square root (HERB)
Percent exotic herbaceous species (EXOTIC)	square root (EXOTIC)
Foliage Height Diversity (FHD)	none

Table 7-3. Total avian abundance and species richness during the breeding season as counted in the different sections of Forest Park and the old-growth stand in the different years of the study.

Category and Year	City	Mid	Far	OG [†]	Total
<u>Avian Abundance</u>					
1993	578	570	540	92	1780
1994	687	678	638	116	2119
1995	683	667	688	125	2163
Total number of birds	1948	1915	1866	333	6062
<u>Avian Species Richness</u>					
1993	43	37	38	21	52
1994	43	38	36	19	48
1995	38	37	39	22	43
Total number of species	52	44	46	30	56

[†] Numbers for the old growth section are for one site whereas numbers for all other sections are for eight sites per section.

Table 7-4. Results of multiple regression analysis for the mean total annual abundance of breeding season birds in the categories of habitat preference, migratory status, and food guild. Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix J.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 [†]
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	
sqrt (# birds)			+	+				0.31**
sqrt (# urban birds)		-					-	0.46***
sqrt (# urban species)							-	0.36***
sqrt (# edge birds)	--							0.41***
sqrt (# forest birds)	+							0.38***
sqrt (# resident birds)			+	+				0.33***
sqrt (# short distance migrants)		-						0.47***
sqrt (# predatory birds)							+	0.29**
sqrt (# ground gleaners)		-				--		0.41***
sqrt (# bark gleaners)				++				0.45***

[†] Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

birds, resident birds, and bark gleaners were positively correlated with the distance from the nearest forest edge, whereas the abundances of urban birds, short-distance migratory birds and ground gleaners were negatively correlated with the distance from the nearest road.

The abundances of several species of birds also correlated with one or more of the urbanization variables (Table 7-5). The abundance of house finches, categorized in this study as urban birds, was positively correlated with the number of surrounding houses. The abundances of spotted towhees and song sparrows, considered to be edge species in this study, correlated negatively with the distance from downtown Portland. The abundance of winter wrens, a species usually associated with forests (although not classified into the forest interior bird category in this study) were positively correlated with the distance from the nearest forest edge.

The old-growth stand had significantly more hawks (includes Cooper's, sharp-shinned, and red-tailed hawks), Steller's jays, red-breasted nuthatches, brown creepers, Swainson's thrushes, and Hutton's vireos (Table 7-6). Many of these bird species are classified as forest birds here and elsewhere. Sites in more urban sections of Forest Park (i.e., the city and middle sections) had significantly more American robins, spotted towhees, and song sparrows (all categorized as either urban or edge species in this study), and significantly fewer Pacific-slope flycatchers (classified as a forest interior species in this study).

There were significantly more birds in the old-growth stand than at any site in Forest Park (Table 7-7). There were also more urban birds, edge birds, and short distance

Table 7-5. Results of multiple regression analysis for the mean total annual abundance of breeding season birds. Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables and the covariable. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix J.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES							R^2 †
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	
sqrt (# hawks††)					---		++	0.46***
sqrt (# Stellers jays)			++				++	0.33***
sqrt (# brown creepers)			+				+	0.32***
sqrt (# winter wrens)				++			-	0.31***
sqrt (# Swainson's thrushes)	+							0.25*
sqrt (# house finches)					+		-	0.31***
sqrt (# spotted towhees)	--	++				-		0.48***
sqrt (# song sparrows)	---							0.31***

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

†† includes Cooper's, sharp-shinned, and red-tailed hawks

Table 7-6. Mean number of breeding season birds (standard deviation) for each section. Bird species that were present in less than two sections or were counted fewer than 30 times over the three year period were considered infrequent or rare species and were omitted from the analyses. n is reported for each section.

<u>Bird Species</u>	<u>City</u>	<u>Mid</u>	<u>Far</u>	<u>OG</u>
Hawks ^{††}	0.5 (1.0) ^a	0.3 (0.6) ^a	0.5 (0.7) ^a	3.0 (2.6) ^b
Mourning doves	0.6 (0.8)	0.2 (0.5)	0.3 (0.6)	0.7 (0.6)
Hairy woodpeckers	0.1 (0.3)	0.5 (1.0)	0.6 (1.0)	1.0 (1.0)
Downy woodpeckers	0.5 (0.8)	0.3 (0.6)	0.5 (1.0)	0.7 (1.2)
Pileated woodpeckers	0.6 (0.8)	0.8 (1.0)	0.9 (1.0)	1.7 (1.5)
Olive-sided flycatchers	0.7 (1.4)	-	0.3 (0.6)	0.7 (0.6)
Pacific-slope flycatchers	4.1 (1.9) ^a	5.9 (1.8) ^b	6.3 (2.0) ^b	7.7 (0.6) ^b
Steller's jays ^x	1.3 (1.5) ^a	1.5 (1.6) ^a	1.5 (1.4) ^a	8.3 (3.2) ^b
Black-capped chickadees	7.7 (7.7)	9.8 (7.2)	6.8 (5.8)	19.7 (14.5)
Chestnut-backed chickadees	1.0 (1.9)	2.7 (3.9)	0.7 (1.3)	1.0 (1.7)
Bush tits	1.4 (3.0)	1.0 (2.2)	0.6 (2.3)	-
Red-breasted nuthatches	3.1 (2.2) ^a	2.4 (2.1) ^a	3.6 (2.1) ^{ab}	7.0 (3.5) ^b
Brown creepers ^x	3.2 (2.0) ^a	3.3 (1.9) ^a	3.6 (1.9) ^a	10.7 (4.6) ^b
Bewick's wrens	0.5 (0.9)	0.1 (0.3)	0.4 (0.8)	-
Winter wrens	7.3 (2.5) ^{ab}	6.4 (2.4) ^a	8.6 (2.0) ^b	10.3 (2.1) ^b
American robins ^x	7.3 (3.2) ^a	4.3 (2.3) ^b	5.3 (2.9) ^{ab}	1.3 (1.5) ^c
Swainson's thrushes ^x	2.7 (1.9) ^a	3.3 (2.1) ^a	3.9 (2.1) ^{ab}	8.3 (5.0) ^b
Golden-crowned kinglets	0.5 (0.9)	0.7 (2.4)	1.1 (1.8)	3.7 (4.0)
Solitary vireos	0.8 (1.1)	0.6 (1.1)	0.4 (0.6)	1.0 (1.7)
Hutton's vireos ^x	0.7 (1.6) ^a	0.3 (0.6) ^a	0.3 (0.6) ^a	3.3 (1.5) ^b
Orange-crowned warblers	4.4 (2.5)	4.6 (2.1)	3.5 (2.7)	1.3 (0.6)
Yellow-rumped warblers	1.5 (5.3)	0.3 (1.1)	0.1 (0.2)	-
Black-throated gray warbler	1.9 (2.1)	4.0 (3.5)	3.4 (2.4)	2.0 (2.6)
Wilson's warblers	5.8 (2.3)	6.2 (2.7)	6.3 (2.7)	5.0 (1.7)
Brown-headed cowbirds	0.2 (0.6)	-	0.2 (0.6)	-
Western tanagers	1.3 (1.6)	1.8 (1.9)	2.7 (1.9)	4.3 (2.3)
Black-headed grosbeaks	1.5 (1.8)	1.3 (1.5)	1.9 (1.4)	0.3 (0.6)
Purple finches	1.9 (2.7)	0.9 (0.9)	1.3 (1.5)	-
House finches ^x	1.0 (2.1) ^a	0.1 (0.2) ^b	0.1 (0.4) ^{ab}	-
Pine siskins	0.6 (1.0)	0.4 (0.7)	0.1 (0.2)	-
Spotted towhees	5.8 (2.1) ^a	4.9 (2.4) ^a	3.7 (2.2) ^b	0.7 (1.2) ^b
Dark-eyed juncos ^x	3.4 (2.1) ^{ab}	4.3 (2.3) ^a	3.8 (1.9) ^{ab}	1.3 (1.2) ^b
Song sparrows ^x	4.0 (2.2) ^a	3.9 (2.5) ^a	1.7 (2.2) ^b	0.0 (0.1) ^b
n	24	24	24	3

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses.

^{††} includes Cooper's, sharp-shinned, and red-tailed hawks

Table 7-7. Mean abundance of total breeding season birds by categories of habitat preference, migratory status, and food guild (standard deviation) per year for each section.

Category	City	Mid	Far	OG
# urban birds ^x	8.6 (4.1) ^a	4.4 (2.2) ^{bc}	5.7 (3.4) ^b	1.3 (1.5) ^c
# urban species ^x	1.6 (0.8) ^a	1.1 (0.4) ^{ab}	1.3 (0.5) ^{ab}	0.7 (0.6) ^b
# edge birds ^x	15.1 (6.7) ^a	10.5 (4.4) ^b	7.7 (3.9) ^b	2.3 (2.1) ^c
# edge species ^x	3.8 (1.2) ^a	3.0 (0.9) ^{ab}	2.9 (1.3) ^{ab}	2.0 (1.7) ^b
# forest birds ^x	13.2 (4.5) ^a	18.0 (4.7) ^b	18.8 (5.6) ^b	31.3 (9.8) ^c
# forest species	4.4 (0.9)	4.7 (1.0)	5.0 (0.6)	5.0 (1.0)
# resident birds	41.8 (16.2) ^a	41.4 (13.7) ^a	39.8 (11.3) ^a	74.0 (23.6) ^b
# neotropical birds	25.5 (7.2)	28.6 (5.5)	28.9 (5.7)	31.7 (8.5)
# short distance migrants ^x	20.2 (6.7) ^a	12.9 (5.3) ^{ab}	13.5 (7.0) ^{ab}	4.0 (2.6) ^c
# predatory birds ^x	0.5 (1.0) ^a	0.5 (0.7) ^a	0.8 (0.9) ^{ab}	3.0 (2.6) ^b
# ground gleaners ^x	33.5 (7.2) ^a	26.9 (5.5) ^b	27.3 (5.8) ^b	24.3 (3.2) ^{ab}
# foliage gleaners ^x	33.0 (10.3) ^a	37.9 (10.8) ^{ab}	32.5 (8.8) ^a	51.7 (11.0) ^b
# bark gleaners ^x	8.0 (3.7) ^a	7.5 (4.0) ^a	9.6 (3.6) ^a	21.0 (6.6) ^b
# aerial gleaners	5.4 (2.2)	6.0 (1.8)	6.7 (2.1)	8.3 (1.2)
# birds ^x	81.2 (15.9) ^a	79.8 (15.0) ^a	77.8 (12.0) ^a	111.0 (17.1) ^b
# species	21.2 (3.6)	20.6 (2.9)	20.6 (1.9)	21.3 (1.2)
range species per site	14-28	16-26	17-24	20-22
H'	2.76 (0.17)	2.74 (0.13)	2.77 (0.11)	2.70 (0.18)
n	24	24	24	3

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses.

migratory birds at sites in the city, whereas there were more forest birds and resident birds in the old-growth. I did not find any significant differences among the sections regarding neotropical species of birds. The old-growth stand, however, had significantly more predatory birds, foliage gleaners, and bark gleaners, whereas the city section had significantly more ground gleaners.

Winter Birds

I observed 2230 individual birds from 35 species during the three winter seasons (Table 7-8). Total bird abundance, abundances of resident birds, and abundances of foliage gleaners were all positively correlated with the old-growth stand (Table 7-9). Several avian variables were also positively correlated with the distance from the center of the forest. Abundances of only two bird species -- brown creepers and golden-crowned kinglets -- were correlated with urbanization variables. The abundance of brown creepers, classified as forest interior species in this study, was positively correlated with the distance from the center of the forest, whereas the abundance of golden-crowned kinglets, classified as forest birds by others, was positively correlated with the old-growth stand.

The abundances of all bird species that were significant (i.e., black-capped chickadees, red-breasted nuthatches, brown creepers, and golden-crowned kinglets) were greater in the old-growth stand (Table 7-10). There were also significantly more forest birds, resident birds, foliage gleaners, and bark gleaners in the old-growth stand than in any section of Forest Park .

Table 7-8. Total avian abundance and species richness as counted in the winter months in the different sections of Forest Park and the old-growth stand during the three years of the study.

Category and Year	City	Mid	Far	OG [†]	Total
<u>Avian Abundance</u>					
1994	138	86	94	66	384
1995	96	110	187	110	503
1996 [‡]	415	352	328	248	1343
Total number of birds	649	548	609	424	2230
<u>Avian Species Richness</u>					
1994	11	8	9	6	17
1995	10	8	13	8	18
1996 [‡]	20	18	20	8	29
Total number of species	26	18	25	11	36

[†] Numbers for the old growth section are for one site whereas numbers for all other sections are for eight sites per section.

[‡] Abundances and richness for 1996 sample represent the total of triplicate sampling whereas only one sample per year was taken for 1994 and 1995.

Table 7-9. Results of multiple regression analysis for the mean winter bird abundance. Results reported as significant positive (+) or negative (-) correlations with each of the seven independent (urbanization) variables. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix J.

DEPENDENT VARIABLE	INDEPENDENT (URBANIZATION) VARIABLES								
	PDX	ROAD	NWI	EDGE	HOUSE	FOREST	CENTER	OG	R ² †
sqrt (# total birds)							++	+++	0.57***
sqrt (# resident birds)							++	+++	0.56***
sqrt (# short distance m.)				-					0.25**
sqrt (# foliage gleaners)								+++	0.45***
sqrt (# bark gleaners)							++		0.39***
sqrt (# BRGR)							+		0.32***
sqrt (# GCKI)								+++	0.38**

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

Table 7-10. Mean abundance of winter birds (standard deviation) for each section. Bird species that were present in less than two sections or were counted fewer than 40 times over the three year period were considered infrequent or rare species and were omitted from the analyses.

	City	Mid	Far	OG
All woodpeckers	0.1 (0.3)	0.2 (0.5)	0.2 (0.5)	0.2 (0.4)
American crows	0.1 (0.3)	0.5 (1.4)	0.1 (0.3)	-
Black-capped chickadees ^x	4.3 (2.5) ^a	3.5 (3.0) ^a	4.1 (4.2) ^a	23.3 (14.7) ^b
Red-breasted nuthatches ^x	0.6 (0.7) ^a	0.4 (0.6) ^a	0.9 (0.7) ^{ab}	2.7 (0.6) ^b
Brown creepers ^x	1.1 (1.1) ^a	1.2 (1.0) ^a	1.0 (0.8) ^a	7.0 (5.0) ^b
Winter wrens	1.6 (1.3)	1.8 (1.3)	2.2 (1.3)	4.4 (0.5)
Golden-crowned kinglets ^x	5.0 (6.2) ^a	3.9 (3.3) ^a	5.8 (8.5) ^a	39.7 (5.5) ^b
Dark-eyed juncos	0.8 (1.2)	0.5 (0.8)	0.5 (0.9)	0.7 (1.2)
# birds ^x	14.8 (7.4) ^a	12.0 (6.3) ^a	15.9 (13.4) ^a	81.3 (25.1) ^a
# species	6.2 (2.8)	5.6 (2.7)	6.4 (2.7)	8.0 (1.0)
range species per site	6-14	6-11	6-13	7-9
H'	1.42 (0.39)	1.40 (0.44)	1.47 (0.37)	1.45 (0.21)
# urban birds	0.1 (0.4)	0.0 (0.1)	0.0 (0.2)	-
# edge birds	0.2 (0.6)	0.1 (0.1)	0.1 (0.2)	-
# forest birds ^x	1.4 (1.2) ^a	1.5 (1.5) ^a	1.3 (0.9) ^a	7.2 (5.0) ^b
# resident birds ^x	14.7 (7.4) ^a	12.5 (6.4) ^a	15.7 (13.5) ^a	81.3 (25.1) ^b
# short distance birds	0.4 (0.8)	0.1 (0.2)	0.2 (0.5)	-
# ground gleaners	3.2 (2.2)	3.0 (2.2)	3.3 (1.6)	5.7 (1.5)
# foliage gleaners ^x	9.9 (6.9) ^a	7.7 (5.9) ^a	10.3 (12.4) ^a	65.3 (18.8) ^b
# bark gleaners ^x	1.8 (1.3) ^a	1.8 (1.3) ^a	2.2 (1.4) ^a	10.0 (5.0) ^b
n	24	24	24	3

Results of Bonferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used in statistical analyses.

In all three years, I observed significantly higher bird abundances at the old-growth stand than at any site in Forest Park (Figure 7-1), although species richness and diversity were similar among the different sections. The increased avian abundances seemed to be positively correlated with snow level in the nearby mountain ranges (i.e., Cascades and Coast Range). To test this hypothesis, I compared the number of birds counted in the old-growth with the snow level at Government Camp (a small town at an elevation of 1190 meters on the south shoulder of nearby Mount Hood) four days earlier using simple regression analysis. Comparing the depth of snow at Government Camp on the same day that the birds were counted would not allow time for the birds to move from higher elevation to the old-growth stand after a substantial snowfall. The regression (Figure 7-2) revealed a significant positive correlation between snow in nearby mountains and bird abundance in the old-growth stand four days later.

Plant Community Structure and Avian Distribution

Several measurements of bird abundance were found to be significantly correlated with some of the plant community variables (Table 7-11). For breeding season birds, the abundances of birds and bark gleaners were negatively correlated with the number of bigleaf maples. The densities of urban birds and short distance migrants, as well as edge species richness, were all negatively correlated with the foliage height diversity (FHD). The abundances of urban birds and short distance migratory species were positively correlated with the basal area of thimbleberry and the percent of exotic species. The abundances of Pileated woodpeckers and Swainson's thrushes (forest species) were both negatively correlated with the density of trees and positively correlated with the foliage height diversity (FHD) (Table 7-12).

Figure 7-1. The mean number of winter birds in Forest Park versus in the old-growth stand for each year of the study.

Figure 7-2. The number of birds in the old growth stand versus snow depth (inches) at Government Camp four days prior to the day that the bird count was conducted. The regression equation is $y = 1.23x + 49.7$; $R^2 = 0.86$; $p=0.02$.

Table 7-11. Results of multiple regression analysis for the mean total annual abundances of breeding season birds in the categories of habitat preference, migratory status, and food guild. Results reported as significant positive (+) or negative (-) correlations with each of the eight independent (plant community) variables. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). For all variables, $n=75$. For regression coefficients, see Appendix K.

DEPENDENT VARIABLE	INDEPENDENT (PLANT COMMUNITY) VARIABLES							R ²	
	TREE	MAPLE	CEDAR	SIRUB	THIMB	I'H	EXOTIC		FID
sqrt (# birds)		--							0.29**
sqrt (# urban birds)					+		++	-	0.28**
sqrt (# edge birds)	-					++			0.50***
sqrt (edge species)								-	0.26**
sqrt (# short distance)					+		+	--	0.35***
sqrt (# bark gleaners)		---							0.35***
sqrt (# predatory birds)							---		0.25*

† Significance level of the squared coefficient of multiple regression; significance levels: * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$
+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p < 0.05$
++ or -- denotes that the dependent variable is significantly correlated with the independent variable with $p < 0.01$
+++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p < 0.001$

Significant correlations with vegetative parameters were different during the winter season than during the breeding season (Table 7-13). The abundances of birds, urban birds, forest birds, resident birds, foliage gleaners, bark gleaners, brown creepers, and golden-crowned kinglets were all negatively correlated with the density of bigleaf maples (a deciduous species), and except for the abundance of urban birds, all the previously listed bird categories were also negatively correlated with the species diversity of the herbaceous layer. In addition, the abundances of urban birds, resident birds, foliage gleaners, and black-capped chickadees were all positively correlated with the density of shrubs.

DISCUSSION

Breeding Bird Community

Several breeding season bird variables correlated with urbanization. Using the distance from downtown Portland as a simple urbanization gradient as described in chapters 5 and 6, I found greater richness and density of birds categorized as edge species, as well as greater density and richness of individual edge species such as spotted towhees and song sparrows at more urban sites. Concomitant with the greater numbers of edge birds, there were fewer forest birds and fewer forest bird species such as winter wrens and Swainson's thrushes. These results are consistent with those from other studies (Askins 1995; Askins and Philbrick 1987; Blair 1996; Blake et al. 1994; Bollinger and Linder 1994; Donovan et al. 1995a; Hoover et al. 1995; Keller and Anderson 1992; Kroodsma 1984; Lent and Capen 1995; McIntyre 1995; McLellan et al. 1986; Stouffer and Bierregaard 1995; Temple and Wilcox

Table 7-13. Results of multiple regression analysis for the mean total annual abundance of winter birds in the categories of habitat preference, migratory status, and food guild. Results reported as significant positive (+) or negative (-) correlations with each of the eight independent (plant community) variables. The squared coefficient of multiple regression (R^2) for each dependent variable is reported as is the significance level (p). For all variables, n=75. For regression coefficients, see Appendix K.

DEPENDENT VARIABLE	INDEPENDENT (PLANT COMMUNITY) VARIABLES							
	TREE	MAPLE	CEDAR	SHRUB	THUMB	H'HI	EXOTIC	R^2 †
sqrt (# birds)		--		+		--		0.36***
sqrt (# urban birds)	+++	-						0.25**
sqrt (# forest birds)		-				--		0.25*
sqrt (# residents)		--		+		--		0.35***
sqrt (# foliage gleaners)		-		+		-		0.31**
sqrt (# bark gleaners)		--				---		0.31**
sqrt (# BCCH)				++				0.26**
sqrt (# BRGR)		--				--		0.27**
sqrt (# GCKI)		-				-		0.27**

† Significance level of the squared coefficient of multiple regression; significance levels: *= $p<0.05$; **= $p<0.01$; ***= $p<0.001$

+ or - denotes that the dependent variable is significantly correlated with the independent variable with $p<0.05$

++ or --- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.01$

+++ or ---- denotes that the dependent variable is significantly correlated with the independent variable with $p<0.001$

1986; Terborgh 1989; Wilcove 1985). However, unlike the previous studies where forested areas were adjacent to developed areas of differing land uses, my findings were confined to a forested area contiguous with a city. The city section of Forest Park has a greater density of edge habitat birds than did the other areas sampled which is consistent with more edge habitat in the city section of the park (see Figure 5-2, page 75).

Many species of birds, the edge species, respond more favorably to increased edge habitat that results from forest fragmentation. Forest bird species, on the other hand, frequently have specialized habitat requirements and are more sensitive to habitat fragmentation (Lent and Capen 1995; Lynch and Whigham 1984; Whitcomb et al. 1981; Wilcove 1985). I found that the richness and abundance of both edge birds and urban birds were greater in edge habitats. Species such as song sparrows, spotted towhees, American robins, and house finches have been associated with edge or suburban areas by others (Blair 1996; Whitcomb et al. 1981) and were also found to be associated with edge habitat in this study. In addition, I found that species classified by others as preferring forest interior habitat such as Cooper's and sharp-shinned hawks, brown creepers, winter wrens, and Hutton's vireos were all associated with forest interior habitat in this study. These results demonstrate that increased development around Forest Park is producing more edge habitat and that the avian community is responding to the changes.

My results show that habitat fragmentation in Forest Park is correlated with changes in the abundance and diversity of birds. But, are these results the consequence of habitat fragmentation or can the observed differences be attributed to different past

land usage patterns in Forest Park? Since forest fires and logging produce a fragmented landscape similar to the effects of development, it may be hard to distinguish between the effects of urban development and other human effects. To investigate the relationships between birds and past land use, I compared the distribution of individual bird species in different sections of the park.

I found significantly more forest birds in the old-growth stand, including Cooper's , sharp-shinned, and red-tailed hawks (nested in the stand), Swainson's thrushes, Steller's jays, red-breasted nuthatches, brown creepers, Hutton's vireos, and winter wrens. Of these species, brown creepers, red-breasted nuthatches, Hutton's vireos, Steller's jays, and winter wrens have all been found to be associated with old-growth and mature forests in the Pacific Northwest in spring (Carey et al. 1991; Gilbert and Allwine 1991; Huff and Raley 1991; Lundquist and Mariani 1991; Manuwal 1991; Ruggiero et al. 1991). While all of these old-growth associated avian species were also found at sites in Forest Park, they were present in reduced abundances as compared to those in the old-growth. I also found significantly more bark gleaners, foliage gleaners, and predatory birds in the old-growth. There have been reports of increased abundance of bark gleaners in old-growth Douglas-fir forests (Carey et al. 1991; Huff and Raley 1991; Lundquist and Mariani 1991). Larger numbers of predatory birds (e.g., Cooper's hawks and Northern pygmy-owls) have been reported in old-growth forests; but due to the large ranges and relative scarcity of the birds, most of these examples have not been statistically significant (Carey et al. 1991, Huff and Raley 1991; Ruggiero et al. 1991). Avian evidence is consistent with the observation that the old-growth stand is significantly different from Forest Park. However, I found significantly reduced density and richness of edge birds in

the old-growth even though the old-growth stand is essentially all edge as a result of the surrounding clear cuts. This finding supports the idea that successional stage, not fragmentation, is the most important factor in avian distribution at this study site. Nevertheless, the increased abundance of all birds may be related to the increased amount of edge habitat surrounding the stand.

The only bird species whose abundances were greatest in the far section were the winter wren and Pacific-slope flycatcher (Table 7-6). Both bird species are typically found in older seral stage forests (Carey et al. 1991; Gilbert and Allwine 1991; Huff and Raley 1991; Manuwal 1991; Ruggiero et al. 1991). I found more forest birds and fewer urban and edge birds at sites in the far section as compared to the city section of Forest Park. In most cases, however, avian abundances in the far and middle sections were not significantly different from one another. Those differences that were observed were most likely the result of the degree of vegetation homogeneity in the different sections. Trees have a more heterogeneous distribution in the far section of Forest Park that is most likely attributable to the more patchy nature of past disturbances (i.e., small scale logging and slash burns in the far section versus a widespread, devastating fire in the middle section). At this time, with the minimal differences in avian distribution between these two sections of Forest Park, it is hard to determine whether fragmentation is playing role in the distribution of birds in either of these two sections.

Avian distribution in the city section was different from that in the other areas. I found greater abundances of both urban and edge birds than in the other sections. I also found greater abundances of such urban-adapted species as American robins and

house finches, as well as more ground gleaners, similar to the findings of others (Aldrich and Coffin 1980; Beissinger and Osborne 1984; Blake 1983; Block et al. 1986; Carey et al. 1991; Davis and Glick 1978; DeGraaf 1991; DeGraaf and Wentworth 1986; Lancaster and Rees 1979; Lent and Capen 1995). In addition, there were increased abundances of such edge-adapted species as spotted towhees and song sparrows. These species have been categorized as edge-adapted species in other studies (Askins and Philbrick 1987; Blair 1996; DeGraaf and Wentworth 1986; Lent and Capen 1995; Levenson 1981; McIntyre 1995; McLellan et al. 1986; Rich et al. 1994; Rosenberg and Raphael 1986; Wilcove 1985). The city section also had significantly fewer forest interior birds, as well as fewer forest associated species such as Pacific-slope flycatchers. The decreased abundance of forest associated species of birds, as well as the increased abundance of urban and edge associated species of birds in the city section supports the conclusion that habitat fragmentation is important in avian patterns of distribution in the city section of the park. In terms of the avian community, the city section of Forest Park appear to be functioning as edge habitat due to the higher degree of habitat fragmentation. In addition, its close proximity to the urban environment appears to be attracting such urban adapted species as house finches.

The increased disturbance regime in and around the city section is likely increasing the patchiness of the landscape, increasing the amount of edge habitat, and providing edge and urban associated bird species a niche that would not normally be present in an older seral forest. It is also possible that not only is the amount of edge important, but so is the matrix or habitat surrounding the forest. Several recent studies have shown that the surrounding habitat and/or land use is more important to the

distribution of birds in a patch than is the size of the patch itself (Friesen et al. 1995; Sisk et al. 1997). The distribution of birds in the old-growth lends some support to this hypothesis as the increased amount of edge habitat surrounding the old-growth stand has not resulted in an increased amount of edge birds, nor a decreased amount of forest birds. Other studies have shown that close proximity to highly traveled roads may result in changes in avian distribution (Reijnen and Foppen 1995). It may be that different land uses surrounding Forest Park, as well as the extensive network of roads in the area, contribute to the effects of edge, especially in the city section of the park where development has, up until the recent past, been far less dramatic.

My avian findings are also supported by correlations with the vegetation parameters. The densities of edge birds were found to be correlated with plant variables in the same manner that the plant variables had been previously correlated with edge habitat. These findings support the contention that the effects of habitat fragmentation are influencing the avian community through vegetative community structure. The effects of habitat fragmentation on the plant community are correspondingly affecting the avian community.

Because neotropical migratory species seem to be especially affected by habitat fragmentation (Friesen et al. 1995; Lynch and Whigham 1984; Sauer et al. 1997; Whitcomb et al. 1981), I expected to see fewer such birds at more urban sites. I found that the abundance of resident bird species declined with urbanization and the abundance of short-distance migratory species increased, similar to the findings of others (Blair 1996; Lent and Capen 1995; Lynch and Whigham 1984; Wilcove 1985). Huff and Raley (1991) reported greater abundance of resident birds in

Douglas-fir old-growth forests as compared to younger seral forests. Although I did find significantly more resident birds in the old-growth, I did not find any significant differences in the distribution of neotropical migratory species. Long-term analyses of results from the Breeding Bird Survey show no significant reduction in the abundance of neotropical migratory birds in the western portions of the United States, in spite of extensive forest fragmentation (Herkert 1995; Peterjohn and Sauer 1994; Sauer et al. 1997). This may be because neotropical migratory birds comprise a smaller percentage of the avifauna in Pacific Northwest forests than they do in eastern forests where most studies have been conducted (Manuwal 1991) or it may be due to other unknown factors.

My results suggest that birds are responding to the successional stage of the forest, but that increased habitat fragmentation also accounts for changes in the avian community near Portland. I found more edge and urban adapted species at sites nearer to the city. Many edge and city adapted species are what Lent and Capen (1995) refer to as "fugitive species". "Fugitive species" are adapted to an ever-changing landscape and are opportunists in nesting and surviving in the ever-changing mosaic of habitats (Lent and Capen 1995). Many Pacific Northwest forest bird species, being generalists, inhabit seral communities from early successional to old-growth forests and lead such a "fugitive" life style. Since "normal" forest structure in the Pacific Northwest is a mosaic of landscapes caused by natural disturbances such as fires and wind storms, it follows that many of the avian species in this area would be well suited to the ever-changing landscape and could exist in a wide range of habitats.

It is possible that certain human disturbances (e.g., logging) mimic certain natural disturbances and have little detrimental effect on the distribution of most Pacific Northwest bird species at the landscape level (McGarigal and McComb 1995; Peart et al. 1992; Raphael and Barrett 1983). However, at some point, urbanization will probably affect the avian community in more insidious ways than through simple habitat fragmentation. I am unaware of any such effects in this study, however one possible example is presence of the brown-headed cowbird in Forest Park. Breeding activity west of the Cascades was first reported in the early 1950s (Gilligan et al. 1994). Although their numbers have grown, an increase in abundance of the species has never been reported in forest fragments generated through logging activity. (Carey et al. 1991; Gilbert and Allwine 1991; Gilligan et al. 1994; Huff and Raley 1991; Lundquist and Mariani 1991; Manuwal 1991; Ruggiero et al. 1991). Because brown-headed cowbirds prefer sites with chronic disturbance and/or pasture-like (i.e., lawn) habitat (Coker and Capen 1995) and because logged lands are usually reforested, their absence from forest "islands" surrounded by logged land is not surprising. Whereas the species was present too infrequently in this study to analyze statistically, it is interesting to note that it was only recorded at sites in the city and far sections of the park -- areas that have large amounts of edge associated with either urban or agricultural development and hence have a greater amount of pasture and lawn (Figure 7-3). Brown-headed cowbirds preferentially parasitize many neotropical bird species (Askins 1995; Rich et al. 1994; Robinson et al. 1995; Whitcomb et al. 1981; Wilcove 1985). Therefore, a decrease in neotropical migratory species at sites near the city could be expected to occur if the abundance of brown-headed cowbirds increases in the Portland area. Concomitantly, the extent of

Figure 7-3. Distribution of brown-headed cowbirds in the study.

predation by both avian (e.g., crows and jays) and mammalian (e.g., raccoons, cats and dogs) predators has also been shown to increase with continued fragmentation and urbanization (Askins 1995; DeGraaf 1995; Hoover et al. 1995; Martin et al. 1996; Rich et al. 1994; Robinson et al. 1995; Wilcove 1985; Yahner and Mahan 1996). Many predators of birds have been shown to prefer neotropical bird species, usually because of their habit of nesting near or on the ground (Bollinger and Linder 1994; Linder and Bollinger 1994; Rich et al. 1994; Wilcove 1985). I did not monitor predation so I cannot determine the effects of predation on avian species in Forest Park. I believe that the increased abundances of certain bird species (e.g., house finches and brown-headed cowbirds) are indicative of the early effects of urbanization and habitat fragmentation on the avian community in Forest Park. However, it is difficult to determine if the bird community is changing, and if so, how it is changing, without detailed temporal studies. Long-term monitoring should be initiated to elucidate any changes that may occur as fragmentation and increased urbanization.

Winter Bird Community

The major difference I found between summer and winter patterns of bird distribution was the greater abundance of birds in the old-growth stand during the winter. I counted significantly more birds, forest birds, black-capped chickadees, red-breasted nuthatches, golden-crowned kinglets, and brown creepers in the old-growth stand than at any site in Forest Park. There were also more bark gleaners, foliage gleaners, and resident birds in the old-growth.

For three consecutive winters (the duration of this study), the old-growth site had far greater abundance of winter resident birds than any of the sites in Forest Park (Figure 7-1). Mean bird abundance at sites in Forest Park ranged from 13.3 to 16.4 birds, whereas the site in the old-growth stand ranged from 66 to 110 birds. For three consecutive years, the old-growth site supported at least five times more birds than any site in Forest Park which suggests that resident birds aggregate in old-growth forests during the winter, particularly in lower elevation old-growth. The data also show that birds in the Pacific Northwest seem to prefer more mature forest in the winter. Morse (1985) reported that although most researchers investigate breeding birds to ascertain the health of avian populations, winter may be the more critical time of the year for the birds, especially for winter residents. Birds that survive the winter go on to breed the following year. The more birds that survive the winter, the better are the chances for increasing population numbers for that species the following breeding season (Graber and Graber 1983; Huff et al. 1991; Manuwal 1991; Mariani 1987; Morse 1985). McIntyre (1995) found increased bird abundance in winter with increased size of unfragmented coniferous habitat in Georgia. Wentworth and Telfer (1993) found increased bird abundance in winter in more mature aspen forests in Alberta. Researchers west of the Cascades have found greater numbers of winter resident birds in old-growth coniferous forests (Huff et al. 1991; Manuwal 1991; Manuwal and Huff 1987, Raphael and Barrett 1983). Old-growth forests exhibit increased structural complexity that is believed to offer superior habitat to wildlife during the winter (Huff et al. 1991; Manuwal 1991; Manuwal and Huff 1987; Mariani 1987; Norse 1990). Old-growth forests have, on average, a much more highly stratified canopy structure than do earlier seral stage forests (Franklin and Spies 1991a; Franklin and Spies 1991b; Spies and Franklin 1991). In this study, the

old-growth stand had a significantly greater foliage height diversity than any site in Forest Park when the ground layer component was not included in the calculation. The increased height and stratification affords animals greater protection from severe climatic conditions and provides more area to forage. These features may be especially important in winter when food supplies are in shorter supply than they are in summer. In addition, old-growth forests west of the Cascades are known to provide a more reliable food source because western hemlocks, on average, produce more cones annually than Douglas-firs (Huff et al. 1991; Manuwal 1991; Manuwal and Huff 1987).

Lower elevation old-growth became increasingly important to birds as snowfall increased in the nearby mountains. Although it is likely that freezing level (or the altitude at which snow accumulation begins) is a more important variable to birds than snow depth at any one location, freezing level is not recorded in the weather archives (George Taylor, Oregon State Climatologist, personal communication) and so could not be studied. More research is needed to clarify the relationship among snowfall, bird abundance, and low elevation old-growth forests.

CHAPTER 8

CONCLUSIONS, FURTHER RESEARCH, AND SUGGESTIONS FOR MANAGEMENT

Generally, the plant and avian communities at my sites are those that would be expected based upon the successional status and past land use history of Forest Park and the old-growth stand. The city section of Forest Park is an exception to this statement. In the city section, there were fewer shade-tolerant trees (many of which are also climax species) than expected given the length of time since the last major disturbance. I found lower densities of saplings of coniferous and shade-tolerant species of trees in the city section than in the other areas of the park. This suggests that the city section appears to be successional younger than would be expected based on time since the last major disturbance. Urbanization appears to have interfered with normal successional progression in the city section of Forest Park to the extent that the city section seems to be in an earlier successional stage than would be expected based upon its age. It appears that the normal successional sequence has been retarded or modified in the this section of the park. This finding is similar to the work of others who have shown that one effect of human disturbance is to "set back" the successional clock (Abrams and Nowacki 1992; Dale and Doyle 1987; Hong et al. 1995; Kowarik 1990; Leak and Smith 1996; Matlack 1993b; Petraitis et al. 1989; Swindel et al. 1986). In the absence of sufficient numbers of young shade-tolerant trees, there will be either continued dominance by hardwoods such as bigleaf maple and/or red alder or, if a potential seed source is available, Douglas-fir could enter and

dominate the forest. Whether the specific causal factors for the shortage of trees are a lack of seed source due to logging and/or large fires or whether they are the result of other human activities will require further study.

The city section was also notable for increased abundances of introduced plant species such as mountain ash, holly and English ivy, as well as greater densities of shrubs, and greater cover by such herbaceous species as bracken fern. The findings are probably the result of the greater levels of edge habitat in the city section of the park. The increase in non-native plant species is an expected consequence of proximity to human development.

Similarly, there was an increase in the abundance of human associated bird species such as house finches and brown-headed cowbirds. Brown-headed cowbirds have been shown to cause significant reductions in the nesting success of many avian species, especially those that nest on or near the ground (Bollinger and Linder 1994; DeGraaf 1995; Leimgruber et al. 1994; Linder and Bollinger 1995; Robinson et al. 1995; Terborgh 1989; Wilcove 1985; Yahner and Mahan 1996). The status of brown-headed cowbirds in Forest Park should be closely monitored in the future. The city section also had increased abundances of urban birds, edge birds, ground gleaning birds, short-distance migrants, American robins, spotted towhees, and song sparrows. In addition, it had lower abundances of forest birds, including Pacific-slope flycatchers and winter wrens. These variations in avian abundance seem to be responses to increasing amounts of edge habitat as indicated by statistical correlations with edge variables and/or levels of disturbance.

I also found that old-growth or mature coniferous forests are important to birds during the winter. The increase in winter bird abundance in the old-growth stand suggests that winter resident species rely upon mature forests during severe weather. Although my sample size was low ($n = 5$), there was a very strong positive correlation between avian abundance in the old-growth stand and severe weather in the surrounding mountain ranges. This finding may be cause for grave concern, considering that very few lower elevation old-growth stands remain, and begs for the preservation of what little is left since we have incomplete understanding of the implications.

FUTURE RESEARCH

My study was exploratory in nature. I was interested in what, if anything, correlated with urbanization in Forest Park. The fact that the city section appears to have been affected by some factor(s) that I believe to be related to urbanization raises the need for further research to identify the causal agent(s). Regular re-examination of my sites may reveal whether or not the observed differences are stable or transient. Possible future research encompasses studying the reasons for the lack of shade-tolerant saplings in the city sections of the park, the introduction and spread of exotic species, the effects of soil compaction in the different sections of the park, the effects of pollutants and/or heavy metals on park populations, and the effects of urbanization upon the soil microfauna in the different sections of the park.

Long term studies that monitor the health and survivorship of trees in the forest will also be important. Such studies will help elucidate whether the differences among sites that I observed are a reflection of increasing urbanization or past land use. Many

upper canopy (dominant) trees in Forest Park are nearing the ends of their life expectancies. Perhaps an opportunity to study age structure and successional progression in Forest Park is available. Such studies will help elucidate successional progression of forested sites that are chronically disturbed by humans.

In terms of avian research, continued monitoring of bird species populations is needed. Long term studies will be critical to understanding the effects of human development on the forest and the associated avifauna. Research could also be conducted on the effects of different levels of disturbance, including that of hikers or bicyclists, on reproductive success by nesting birds. Although I did find some differences in avian abundance, avian abundance is not always a good indication of reproductive success. It appears that small forest remnants in the Eastern and Midwestern sections of the country act as avian sinks by drawing in birds from larger forested tracts to reproduce, although these birds produce few offspring (Brown and Robinson 1996; Donovan et al. 1995; Hoover et al. 1995; Robinson et al. 1995). Research should be done to determine whether and/or to what extent Forest Park is acting as an avian sink. To help accomplish this, work needs to be done on the effects of predation and parasitism on nesting birds in Forest Park, as well as measurements of reproductive success and avian return rates.

SUGGESTIONS FOR MANAGEMENT

Given the management objective that Forest Park proceed along a normal successional sequence to an "old-growth forest", I have the following suggestions. First, the reduced abundance of shade-tolerant saplings implies that the city section will not proceed along a traditional successional pathway without intervention. I

recommend planting native shade-tolerant tree saplings in the city section. Subsequently, the saplings should be monitored for survivorship. If the reduction in shade-tolerant saplings is the result of decreased recruitment or a lack of shade-tolerant tree seed sources, merely planting the trees will eventually allow for overstory development of climax species. However, if the reduction is caused by some other factor of urbanization (e.g., increased herbivore pressure, or increased mortality of seedlings and saplings due to pollutants, trampling, soil compaction, etc.), more intensive intervention may be required, or acceptance of a modified seral state must be considered.

It is also important for the city to fight the further introduction and spread of non-native plant species as much as possible. Forest Park's "No Ivy League" already is making great strides against ivy in the park, although the work needs to continue. Removing ivy is a continual battle that can not be won as long as so much land surrounding the park (much of it in private ownership) is covered by ivy. However, continued vigilance can help reduce the spread of ivy into the forest interior. Removal of exotic species, however, should not stop with English ivy, but should be expanded to include many other introduced species that appear to be spreading throughout the park such as holly, honesty (*Lunaria annua* L.), Himalayan blackberry, and nipplewort. In addition, the city should continue to work with neighboring developments and neighborhood associations to educate people about the problems of certain invasive plant species and encourage the use of native plants for landscaping.

The spread of the brown-headed cowbird into the park should be closely monitored. Although their numbers are very small at present, it may, at some point in the future, become necessary to take steps to limit encroachment by this species. Other avian species also need to be monitored. Human adapted species, such as house finches, that are known to reside inside the park, may outcompete forest interior species and adversely affect native population numbers. Other human adapted and exotic species (e.g., rock doves, house sparrows and European starlings) were not recorded inside of the park by this study. Since many of these species are associated with edge and human development, perhaps the single most important control would be to limit the creation of additional edge and large open expanses (e.g., lawns) near the park.

It is very important for Portland Parks and Recreation to protect forest interior habitat. Both the middle and far sections of Forest Park contain significant amounts of forest interior habitat (figure 5-1, page 73). The Natural Resources Management Plan (Portland Parks and Recreation 1995) stressed the need to protect wildlife habitat in the park, especially in the far (north unit) and middle (central unit) sections of the park. The city section has the greatest human usage at present, although usage has increased dramatically in the middle section in the last decade or so (Portland Parks and Recreation 1995). I encourage the city to continue to plan for better access in the city section, but to limit access points into the middle and far sections. My results indicate that urbanization and development are "retarding" normal successional processes in the city section of the park. As development encroaches on all sides of the park, the surrounding lands will become more fragmented. The middle and far sections, although having more forest interior habitat, already have a high percentage of edge habitat (figure 5-2, page 74). Further fragmentation of lands

surrounding the park could alter the normal successional processes in these sections, too. To minimize this problem, land surrounding the park, especially around the middle and far sections, should be purchased when possible so that it can be preserved rather than developed. Continued development, especially along the southwest boundary of the park, will cause increased habitat fragmentation and degradation of wildlife habitat. There will also be changes as lands to the northwest continue to be cleared and developed. Metro has purchased some of the clearcut land in this area for preservation. However, continued loss of lands to development could isolate Forest Park which would ultimately impact wildlife populations and probably eliminate wide ranging species such as bear, mountain lion, and elk. Long term monitoring of the plant and animal communities in and around the park will be important.

The future of the old-growth stand is also of concern. As the site is developed and a hiking trail is constructing up to and into the area of oldest trees (i.e., the forest interior), the integrity of the forest will be compromised. The Ancient Forest Preserve Master Plan is set up so that the interior trail is limited in scope and much of the forest will not be involved (Ancient Forest Preserve Master Plan 1996). However, only continual monitoring will reveal whether and how much the forest is altered as the usage by humans increases. In this regard, baseline data are urgently needed. Of special importance is the refuge that the old-growth forest system provides birds during winter. As such, avian use in winter, particularly after storms, should be monitored. It may be necessary to limit human access to the site in winter months to avoid disturbing the avifauna. Because human usage will most likely be lower in the winter months, such restrictions may be unnecessary. Of added interest

is the fact that a pair of bald eagles was observed roosting at the site last spring (Lee Kellogg, Friends of Forest Park, personal communication, March 1998). The usage of the site by bald eagles, a threatened species, could require specific restrictions.

Most of the land around the old growth stand is zoned commercial forest or rural residential and several properties have conservation easements. The Ancient Forest Preserve Master Plan (1996) determines to work with adjacent property owners to establish a buffer around the site to minimize the edge effect. However, as land becomes increasingly scarce in the Portland area, pressures to develop some of the surrounding property may increase. To maintain the integrity of the forest, as much of the land as possible surrounding the forest stand should be purchased and preserved.

My research has revealed some effects of urbanization and human development upon forested lands surrounding Portland, Oregon. Overall, the minimization of edge habitat is a key to maintaining a "healthy" forest system. Long term monitoring will help elucidate whether my observations are stable or transient, as well as provide information as to changes that occur with increasing human disturbance. Regardless, this study has provided valuable baseline data to which future data can be compared.

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APPENDIX A

SCIENTIFIC AND COMMON NAMES OF SPECIES OF TREES OBSERVED IN
FOREST PARK AND THE OLD-GROWTH STAND

SCIENTIFIC AND COMMON NAMES OF SPECIES OF TREES OBSERVED IN
FOREST PARK AND THE OLD-GROWTH STAND

(* = non-native)

SCIENTIFIC NAME (AUTHORITY)	COMMON NAME
<i>Abies grandis</i> (Douglas)	Grand fir
<i>Acer macrophyllum</i> (Pursh)	Bigleaf maple
<i>Alnus rubra</i> (Bong)	Red alder
<i>Amelanchier alnifolia</i> (Nuttall)	Serviceberry
<i>Arbutus menziesii</i> (Pursh)	Pacific madrone
<i>Chamaecyparis lawsoniana</i> (Parl)	Port Orford cedar*
<i>Cornus nuttallii</i> (Aud)	Pacific dogwood
<i>Crataegus monogyna</i> (Jacq)	Hawthorn*
<i>Euonymus occidentalis</i> (Nuttall)	Western wahoo
<i>Populus trichocarpa</i> (T and G)	Black cottonwood
<i>Prunus emarginata</i> (Douglas Walp)	Bitter cherry
<i>Pseudotsuga mensiezi</i> (Mirbel Franco)	Douglas-fir
<i>Rhamnus purshiana</i> (DC)	Cascara
<i>Sorbus aucuparia</i> (Linneaus)	Mountain ash*
<i>Taxus brevifolia</i> (Nuttall)	Pacific yew
<i>Thuja plicata</i> (Donn.)	Western red cedar
<i>Tsuga heterophylla</i> (Raf Sarg)	Western hemlock

Nomenclature from Hitchcock and Cronquist 1974 except for *Chamaecyparis lawsoniana* which came from Peck 1941

APPENDIX B

SCIENTIFIC AND COMMON NAMES OF SPECIES OF SHRUBS OBSERVED
IN FOREST PARK AND THE OLD-GROWTH STAND

SCIENTIFIC AND COMMON NAMES OF SPECIES OF SHRUBS OBSERVED
IN FOREST PARK AND THE OLD-GROWTH STAND

(* = non-native)

SCIENTIFIC NAME (AUTHORITY)	COMMON NAME
<i>Acer circinatum</i> (Pursh)	Vine maple
<i>Corylus cornuta</i> (Marsh)	Filbert
<i>Holodiscus discolor</i> (Pursh Maxim)	Oceanspray
<i>Ilex aquifolium</i> (Linneaus)	English holly*
<i>Oemleria cerasiformis</i> (T and G. Greene)	Indian plum
<i>Philadelphus lewisii</i> (Pursh)	Mock orange
<i>Rosa gymnocarpa</i> (Nuttall)	Little wood rose
<i>Rubus discolor</i> (Weihe and Nees)	Himalayan blackberry*
<i>Rubus parviflorus</i> (Nuttall)	Thimbleberry
<i>Rubus spectabilis</i> (Pursh)	Salmonberry
<i>Sambucus racemosa</i> (Linneaus)	Red elderberry
<i>Symphoricarpos albus</i> (L. Blake)	Snowberry
<i>Vaccinium parvifolium</i> (Smith)	Red huckleberry
<i>Prunus laurocerasus</i> (Linneaus)	English laurel*

Nomenclature from Hitchcock and Cronquist 1974 except for *Ilex aquifolium* and *Prunus laurocerasus* which are from Bailey and Bailey 1976.

APPENDIX C

SCIENTIFIC AND COMMON NAMES OF SPECIES OF HERBACEOUS
PLANTS OBSERVED IN FOREST PARK AND THE OLD-GROWTH STAND

SCIENTIFIC AND COMMON NAMES OF SPECIES OF HERBACEOUS
PLANTS OBSERVED IN FOREST PARK AND THE OLD-GROWTH STAND

(* = non-native; *? = questionable origin)

SCIENTIFIC NAME (AUTHORITY)	COMMON NAME
<i>Achlys triphylla</i> (Smith D.C.)	Vanilla leaf
<i>Actaea rubra</i> (Ait)	Baneberry
<i>Adenocaulon bicolor</i> (Hook)	Pathfinder
<i>Adiantum pedatum</i> (Linneaus)	Maidenhair fern
<i>Anemone deltoidea</i> (Hook)	Western anemone
<i>Arenaria macrophylla</i> (Hook)	Bigleaf sandwort
<i>Asarum caudatum</i> (Lindl)	Wild ginger
<i>Athyrium filix-femina</i> (Linneaus, Roth)	Lady fern
<i>Berberis nervosa</i> (Pursh)	Oregon grape
<i>Campanula scouleri</i> (Hook)	Bellflower
<i>Cimicifuga elata</i> (Nutt)	Bugbane
<i>Circaea alpina</i> (Linneaus)	Enchanter's nightshade
<i>Clematis ligusticifolia</i> (Nutt)	Western clematis*?
<i>Corallorhiza</i> sp. (Chat)	Coral-root orchid
<i>Disporum hookeri</i> (Torr. Nicholson)	Fairy-bells
<i>Dryopteris austriaca</i> (Jacq Woynar)	Wood fern
<i>Epilobium minutum</i> (Lindl)	Willow weed
<i>Galium aparine</i> (Linneaus)	Bedstraw, velcro plant
<i>Gaultheria shallon</i> (Pursh)	Salal
<i>Geranium robertianum</i> (Linneaus)	Herb Robert*
<i>Hedera helix</i> (Linneaus)	English ivy*
<i>Heuchera micrantha</i> (Douglas)	Smallflower alumroot
<i>Hydrophyllum tenuipes</i> (Heller)	Pacific waterleaf
<i>Lactuca muralis</i> (L. Fresen)	Wall lettuce*
<i>Lapsana communis</i> (Linneaus)	Nipplewort*
<i>Lonicera ciliosa</i> (Pursh)	Orange honeysuckle
<i>Melica bulbosa</i> (Geyer)	Oniongrass
<i>Melissa officinalis</i> (Linneaus)	Lemon balm*
<i>Montia sibirica</i> (L. Howell)	Candy flower
<i>Nemophila parviflora</i> (Douglas)	Small-flowered nemophila
<i>Osmorhiza chilensis</i> (H. and A)	Sweet-cicely
<i>Oxalis corniculata</i> (Linneaus)	Yellow oxalis*
<i>Petasites frigidus</i> (Linneaus Fires)	Coltsfoot
<i>Phalaris arundinaceae</i> (Linneaus)	Reed canarygrass*?
<i>Polypodium glycyrrhiza</i> (D.C. Eat)	Licorice fern
<i>Polystichum lonchitis</i> (L. Roth)	Christmas fern*
<i>Polystichum munitum</i> (Kaulf Presl)	Sword fern
<i>Pteridium aquilinum</i> (L. Kuhn)	Bracken fern
<i>Rubus leucodermis</i> (Douglas)	Blackcap
<i>Rubus ursinus</i> (Cham. and Schlecht.)	Wild blackberry

SCIENTIFIC NAME (AUTHORITY)	COMMON NAME
<i>Senecio vulgaris</i> (Linneaus)	Common groundsel*
<i>Smilacina racemosa</i> (L. Desf)	False Solomon's seal
<i>Smilacina stellata</i> (L. Desf.)	Star-flowered Solomon's seal
<i>Stachys cooleyae</i> (Heller)	Cooley's hedge-nettle
<i>Stellaria crispa</i> (Cham & Schlecht)	Starwort
<i>Streptopus amplexifolius</i> (Linneaus)	Twisted stalk
<i>Tellima grandiflorum</i> (Pursh Dougl)	Fringecup
<i>Tiarella trifoliata</i> (Linneaus)	Foamflower
<i>Trientalis latifolia</i> (Hook)	Starflower
<i>Trillium ovatum</i> (Pursh)	Trillium
<i>Vancouveria hexandra</i> (Hook; Morr and Dec)	Inside-out-flower, duckfoot
<i>Vicia hirsuta</i> (L. SF Gray)	Hairy vetch*
<i>Viola glabella</i> (Nutt)	Yellow wood violet

PRESENT AT SITES -- NOT ON TRANSECTS

SCIENTIFIC NAME (AUTHORITY)	COMMON NAME
<i>Aquilegia formosa</i> (Fisch)	Red columbine
<i>Disporum smithii</i> (Hook Piper)	Fairy lanterns
<i>Geum macrophyllum</i> (Willd)	Large-leaved avens
<i>Lilium columbianum</i> (Hanson)	Tiger lily
<i>Linnaea borealis</i> (Linneaus)	Twin flower
<i>Montia perfoliata</i> (Donn, Howell)	Miner's lettuce
<i>Monotropa unifora</i> (Linneaus)	Indian pipe
<i>Prunella vulgaris</i> (Linneaus)	Self heal
<i>Rumex crispus</i> (Linneaus)	Curly dock
<i>Tolmiea menziesii</i> (Pursh; T and G)	Pig-a-back plant

Nomenclature from Hitchcock and Cronquist 1974 except *Polystichum lonchitis* which is from Bailey and Bailey 1976.

APPENDIX D

REGRESSION COEFFICIENTS FOR PLANT COMMUNITY
CHARACTERISTICS

REGRESSION COEFFICIENTS FOR HUMAN USAGE AND PLANT COMMUNITY CHARACTERISTICS

Tree abundance, species composition, and species diversity

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt(# trees)	5.99	0.21	0.36	-0.54	-3.54	-1.88	-0.99	0.24	-0.03
ln(# hemlock+1)	2.29	0.29	-0.08	0.03	-1.94	-1.11	-1.63	0.41	0.67
sqrt(# alder)	2.34	0.11	0.86	0.67	-4.30	-2.37	-0.07	0.01	-2.72
sqrt(# cedar)	2.05	0.30	-1.06	-0.81	-2.96	-3.60	0.63	0.22	1.05
ln(# shade tolerant+1)	3.00	0.04	0.45	0.02	-1.14	-0.34	-0.17	0.02	-0.91
# deciduous trees	24.56	0.54	7.78	4.71	-23.73	-13.13	2.64	0.42	-9.41
sqrt(# conifers)	2.30	0.29	-0.08	0.03	-1.94	-1.11	-1.63	0.41	0.67
# tree species	4.26	0.37	1.50	-0.08	-5.70	-5.70	1.43	0.53	-0.76
H' trees	1.21	0.06	0.36	-0.37	-0.63	-0.75	0.06	0.12	0.22

Tree Diameter (dbh in cm)

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt (total dbh all trees)	23.80	0.49	1.24	2.22	-7.32	2.05	-4.25	0.65	1.35
sqrt (total dbh alder)	11.37	0.39	4.71	4.22	-19.64	-6.86	-2.02	-0.15	-15.79
sqrt (total dbh cascara)	1.57	0.03	0.07	-1.55	-2.10	-1.47	2.22	-0.09	1.53
sqrt total dbh cedar	11.49	1.37	-6.81	-6.66	-7.70	-15.77	-6.14	1.11	8.14
sqrt (total dbh tolerant)	8.01	1.35	-8.05	-2.90	-4.16	-8.32	-5.76	1.03	7.42
sqrt (total dbh conifers)	5.63	0.70	0.44	2.11	7.70	2.23	-9.46	1.78	10.78
ln (mean dbh Doug +1)	-0.81	-0.27	0.65	7.92	5.92	-1.77	3.40	0.08	-1.13
sqrt (mean dbh cascara)	0.76	0.02	0.30	-0.89	-1.04	0.73	0.60	0.06	1.14

Tree Importance Values (IV)

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt (IV maple)	0.69	-0.01	-0.12	0.51	-0.34	0.08	0.39	-0.09	-0.97
sqrt (IV cedar)	0.58	0.09	-0.35	-0.23	-0.81	-1.07	-0.19	0.07	0.24
sqrt (IV conifers)	0.41	0.04	0.08	0.07	0.28	0.03	-0.37	0.10	0.50
sqrt (IV shade tolerant)	0.35	0.08	-0.41	-0.06	-0.47	-0.65	-0.05	0.06	0.29
sqrt (IV deciduous)	1.02	-0.02	0.06	0.30	-0.38	-0.03	0.45	-0.07	-0.84

Shrub Species Abundance, Composition, and Diversity, Total Shrub Height and Basal Area, and Shrub Importance Values

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
ln (# shrubs + 1)	3.04	-0.16	0.15	-0.19	-0.37	0.57	-0.22	0.03	1.23
sqrt (H' shrubs)	0.57	-0.05	0.58	0.04	-0.36	0.46	0.07	-0.02	-0.15
sqrt (# non-native shrubs)	0.80	-0.07	0.67	-0.93	0.98	0.32	-0.96	0.20	1.06
sqrt (# red elderberry)	0.31	-0.13	-1.41	2.27	-0.78	0.27	0.51	-0.63	-2.22
sqrt (# thimbleberry)	2.76	-0.17	0.93	-1.70	-0.53	3.00	-1.11	-0.26	1.00
sqrt (# vine maple)	-0.96	0.13	-0.72	2.13	-3.86	-2.92	1.97	0.19	-0.50
sqrt (total shrub height)	9.93	-0.53	0.44	-0.23	-1.57	0.36	-0.78	0.47	4.22
sqrt (IV thimbleberry)	1.30	-0.09	0.53	-0.74	-0.57	1.58	-0.40	-0.16	0.34
sqrt (IV vine maple)	-0.59	0.09	-0.33	1.09	-2.08	-1.72	1.04	0.12	-0.57

Percent Herbaceous Cover

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt (bracken fern)	0.34	-0.01	-0.01	-0.23	-0.26	-0.08	-0.01	-0.01	-0.25
sqrt (fairy bells)	0.30	-0.02	0.03	-0.24	0.44	-0.08	-0.03	0.03	0.31
sqrt (S. stellata)	0.29	-0.02	0.08	0.11	-0.08	-0.13	-0.14	0.01	-0.12
sqrt (sword fern)	-0.06	0.00	0.10	0.26	-0.04	0.16	0.43	-0.03	-0.30
sqrt (vancouveria)	0.50	-0.03	0.45	-0.28	0.50	-0.28	-0.17	0.01	0.32
sqrt (violet)	0.21	-0.02	0.25	0.03	0.04	-0.14	-0.06	0.02	-0.08
sqrt (wild blackberry)	0.75	-0.03	0.24	-0.29	-0.11	-0.05	-0.26	-0.02	0.25
sqrt (bare ground)	0.30	0.04	-0.31	0.08	-0.01	-0.09	0.11	-0.02	-0.06
ln (# species + 1)	3.36	-0.08	0.44	-0.02	-0.02	-0.06	-0.21	0.03	0.64
(H')	1.68	-0.04	0.23	-0.22	0.05	-0.22	-0.30	0.01	0.14
sqrt (# native species)	5.10	-0.13	0.64	-1.14	0.19	-0.16	-0.62	0.04	1.19
sqrt (H' native herbs)	2.01	-0.04	0.27	-0.29	0.10	-0.23	-0.32	0.01	0.22

Foliage Height Diversity (FHD)

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
% cover cedar 10m	0.09	0.01	-0.08	-0.10	0.12	0.02	-0.14	0.00	0.48
FD 10m	0.67	-0.01	0.44	-0.10	-0.52	0.03	-0.34	0.02	0.71
% cover cedar 5m	0.08	0.02	-0.16	-0.12	0.14	0.06	-0.23	0.01	0.48
FD 5m	0.51	0.03	-0.05	0.02	-0.59	-0.03	-0.12	-0.03	0.46
sqrt (% sky 3m)	1.09	-0.02	0.36	-0.20	0.84	-0.38	-0.05	-0.02	0.03
FD 3m	0.09	0.01	-0.45	0.06	-1.20	0.58	0.06	0.01	0.03
FD ground	1.07	-0.07	1.00	0.62	0.42	-0.62	0.13	-0.07	-0.48
Total FHD	4.93	-0.03	0.61	0.93	-2.61	-0.39	-1.44	-0.07	-0.14

APPENDIX E

REGRESSION COEFFICIENTS FOR TREE POPULATION DEMOGRAPHICS -
RECRUITMENT, SIZE CLASS STRUCTURE, MORTALITY, GROWTH RATE,
AND LIVE CROWN RATIO (LCR)

**REGRESSION COEFFICIENTS FOR TREE POPULATION DEMOGRAPHICS - RECRUITMENT, SIZE
CLASS STRUCTURE, MORTALITY, GROWTH RATE, AND LIVE CROWN RATIO (LCR)**

Tree Recruitment and Size Class Structure

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt (# saplings 5-10)	2.75	0.18	0.04	-1.82	-1.64	0.22	-2.32	0.03	0.83
sqrt (# conifers <5)	0.18	0.14	0.67	1.67	-3.43	-1.86	0.17	0.15	-0.55
sqrt (# conifers 5-10)	1.32	0.21	-0.37	-0.58	-2.11	0.14	-1.25	0.18	-0.37
sqrt (# shade tol <5)	0.98	0.19	-0.53	-0.14	-1.65	-2.32	-0.15	0.19	1.49
sqrt (# shade tol 5-10)	0.88	0.21	-1.30	-1.09	-0.12	-0.47	-0.60	0.08	1.07
sqrt (# shade tol 10-20)	0.82	0.12	-1.66	0.13	0.07	-0.90	-0.50	0.03	0.80
sqrt (# shade tol 40-60)	0.60	0.08	-0.41	-0.39	0.20	-0.54	-0.71	0.03	0.86
sqrt (# shade intol 10-20)	1.16	0.17	1.06	-0.20	-1.45	-0.76	-0.54	0.03	-1.44
sqrt (# shade intol >60)	-0.44	-0.12	1.00	0.78	1.32	2.08	0.08	-0.06	0.92
sqrt (# cascara sap<5)	1.73	0.05	0.64	-2.94	0.16	-1.53	1.53	0.14	2.91
sqrt (# hemlock <5)	-0.77	0.04	-0.22	0.23	1.73	0.44	-0.10	0.06	1.40
sqrt (# cedar<5)	1.95	0.19	-0.74	-0.41	-3.29	-2.87	-0.38	0.13	0.59
sqrt (# cedar 5-10)	0.92	0.17	-0.90	-0.61	-1.28	-1.13	-0.44	0.13	0.63
sqrt (# cedar 10-20)	1.90	0.13	-1.53	-0.71	-0.32	-2.11	-1.17	0.06	1.40
sqrt (# alders 20-40)	2.53	0.11	0.13	0.02	-3.19	-2.61	-0.44	0.08	-1.56

Tree Mortality

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt (# dead hemlock)	-0.05	0.11	-1.05	-0.88	2.10	2.08	-0.39	-0.14	1.00
sqrt (# dead cedar)	2.05	0.16	-1.37	-1.41	0.16	-1.17	-1.90	0.12	1.29
sqrt (# dead trees 1996)	2.77	0.25	-0.08	-1.04	0.50	0.06	-1.44	0.13	-0.74
sqrt (# dead conifers)	1.77	0.21	-1.29	-2.40	2.69	2.31	-2.54	0.08	1.85
sqrt (# shade tolerant)	1.41	0.22	-1.86	-1.83	1.74	1.13	-1.55	-0.04	1.65
Mortality rate	0.51	0.02	-0.02	-0.20	0.29	0.07	-0.14	0.01	0.87

Growth rate (cm/year) for the different decades by species

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt (cedar 10-20yr)	-0.87	0.46	1.09	-2.60	-3.14	1.14	-0.08	-0.08	0.38
sqrt (grand fir 10-20yr) [†]	3.88	-0.11	1.39	-5.21	5.98	-0.09	-2.13	0.57	-
sqrt (hemlock 20-30yr)	1.82	0.08	1.08	-2.93	-1.55	1.15	-0.45	0.32	2.29
sqrt (cedar 20-30yr)	1.47	0.15	0.91	-1.15	0.39	0.22	-1.08	0.15	0.44
sqrt (grand fir 20-30yr) [†]	3.01	-0.04	1.60	-2.97	3.24	-0.45	-1.19	0.34	-
sqrt (hemlock 30-40yr)	-1.79	0.07	0.87	0.29	-2.62	2.03	1.43	0.14	-0.14
sqrt (Doug fir 40-50yr)	1.75	0.01	-0.53	-0.51	-0.60	0.59	-0.61	0.02	0.18
sqrt (maple 50-60yr)	0.87	0.02	0.69	0.38	-0.44	0.05	0.38	0.02	-0.91
sqrt (alder 50-60yr) [†]	2.87	-0.06	1.24	-0.95	1.77	-1.30	0.20	0.07	-
sqrt (Doug fir 50-60yr)	1.20	0.02	-0.10	-0.16	-0.49	0.40	-0.20	0.03	-0.14
sqrt (Doug fir 60-70yr)	1.12	0.02	-0.24	0.19	-0.51	0.33	-0.27	0.00	-0.55

[†] regression analysis including covariable could not be run -- no specimens in old growth stand

Live Crown Ratio (LCR)

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
LCR grand fir	1.48	-0.08	0.74	-1.56	2.13	-0.19	-0.57	0.13	-
LCR up canopy maple	0.28	-0.02	0.30	0.61	-0.88	0.02	0.27	-0.04	-
LCR up canopy doug fir	1.01	0.00	0.18	-0.04	-0.19	-0.11	-0.13	-0.01	-0.19
LCR up canopy intol trees	0.56	-0.01	0.21	0.35	-0.26	-0.07	0.13	-0.02	-0.33

APPENDIX F

THE MEAN NUMBER OF SAPLINGS AND TREES BY SIZE CLASS
CATEGORY IN THE DIFFERENT SECTIONS

THE MEAN NUMBER OF SAPLINGS AND TREES (Standard Deviation) BY
SIZE CLASS CATEGORY IN THE DIFFERENT SECTIONS

Size Categories/Species	City	Mid	Far	OG
<u>Saplings <5cm dbh:</u>				
BL maple saplings <5cm dbh ^x	1.3 (2.0) ^{ab}	1.9 (3.3) ^a	0.2 (0.6) ^b	0.0 (-) ^a
Cascara saplings <5cm dbh	1.5 (2.9)	0.6 (1.1)	1.5 (2.2)	0.3 (0.6)
W.Hemlock saplings <5cm dbh ^x	0.5 (1.1) ^a	0.0 (0.2) ^{ac}	0.8 (1.2) ^{ad}	4.7 (2.5) ^b
WRCedar saplings <5cm dbh ^x	0.0 (-) ^a	1.0 (2.2) ^{ab}	1.5 (1.7) ^b	3.7 (3.5) ^b
Grand fir <5cm dbh	0.7 (2.3)	0.3 (0.9)	0.3 (1.0)	0.0 (-)
Saplings all species <5cm dbh	4.3 (4.0)	5.4 (6.2)	5.0 (2.7)	9.0 (2.6)
Conifer saplings <5cm dbh ^x	1.2 (2.4) ^a	2.4 (3.2) ^{ac}	2.8 (2.6) ^{bc}	8.3 (3.2) ^b
Deciduous saplings <5cm dbh	3.1 (3.8)	3.1 (4.6)	2.2 (2.6)	.7 (0.6)
Shade tolerant <5cm dbh ^x	0.5 (1.1) ^a	1.1 (2.3) ^a	2.5 (2.2) ^{bc}	8.3 (3.2) ^{bd}
Shade intolerant <5cm dbh ^x	1.3 (2.0) ^{ab}	2.6 (3.9) ^a	0.3 (0.6) ^b	0.0 (-) ^{ab}
<u>Sapling 5-10cm dbh:</u>				
BL maple saplings 5-10cm dbh	0.2 (0.4)	0.5 (1.3)	0.4 (1.1)	0.0 (-)
WHemlock saplings 5-10cm dbh ^x	0.0 (0.2) ^a	0.0 (0.2) ^a	0.6 (1.0) ^b	2.0 (3.5) ^{ab}
WRCedar saplings 5-10cm dbh ^x	0.0 (-) ^a	0.1 (0.4) ^a	1.3 (2.8) ^b	3.3 (4.9) ^b
Douglas-fir 5-10cm dbh	0.0 (-)	0.5 (0.8)	0.3 (1.1)	0.0 (-)
Grand fir 5-10cm dbh	1.2 (3.2)	0.0 (0.2)	0.5 (1.2)	0.0 (-)
Saplings all species 5-10cm dbh	1.7 (3.1)	1.7 (2.4)	3.3 (3.6)	6.3 (5.5)
Conifer saplings 5-10cm dbh ^x	1.2 (3.2) ^a	0.9 (1.4) ^{ab}	2.8 (3.0) ^b	5.3 (4.7) ^{ab}
Deciduous saplings 5-10cm dbh	0.5 (0.7)	0.8 (1.9)	0.5 (1.1)	1.0 (1.7)
Shade tolerant 5-10cm dbh ^x	0.1 (0.2) ^a	0.2 (0.6) ^a	2.0 (2.8) ^b	5.3 (4.7) ^b
Shade intolerant 5-10cm dbh	0.2 (0.4)	1.1 (1.6)	0.8 (1.5)	0.0 (-)
<u>Trees 10-20cm dbh:</u>				
BL maple trees 10-20cm dbh ^x	0.9 (1.3) ^a	2.4 (2.8) ^{ab}	3.1 (3.4) ^b	1.3 (1.5) ^{ab}
RAlder trees 10-20cm dbh	0.0 (0.2) ^a	1.0 (1.6) ^b	0.3 (0.5) ^{ab}	0.0 (-) ^{ab}
WHemlock trees 10-20cm dbh	0.8 (1.5)	0.0 (0.2)	0.7 (1.3)	1.7 (1.5)
WRCedar trees 10-20cm dbh ^x	0.0 (-) ^a	0.1 (0.3) ^a	1.2 (1.8) ^b	3.0 (2.6) ^b
Douglas-fir 10-20cm dbh	0.1 (0.3)	0.4 (1.1)	1.0 (1.8)	0.0 (-)
Grand fir 10-20cm dbh	0.5 (1.6)	0.2 (0.5)	0.3 (0.7)	0.0 (-)
Trees all species 10-20cm dbh	2.5 (2.5) ^a	4.5 (3.3) ^{ab}	6.5 (4.6) ^b	6.3 (3.2) ^{ab}
Conifer trees 10-20cm dbh	1.4 (2.2) ^a	0.8 (1.5) ^a	3.0 (2.7) ^b	4.7 (3.1) ^b
Deciduous trees 10-20cm dbh ^x	1.1 (1.5) ^a	3.7 (3.2) ^b	3.4 (3.6) ^b	1.7 (1.5) ^{ab}
Shade tolerant 10-20cm dbh ^x	0.8 (1.5) ^a	0.2 (0.5) ^a	1.8 (2.4) ^b	4.7 (3.1) ^b
Shade intolerant 10-20cm dbh ^x	1.0 (1.4) ^a	3.8 (3.6) ^b	4.3 (4.2) ^b	1.3 (1.5) ^{ab}

Size Categories/Species	City	Mid	Far	OG
<u>Trees 20-40cm dbh:</u>				
BL maple trees 20-40cm dbh	4.7 (3.6)	4.5(3.4)	5.5 (3.3)	1.7 (0.6)
RAlder trees 20-40cm dbh ^x	1.8 (2.8) ^a	3.1 (2.0) ^b	2.5 (2.7) ^{ab}	0.0 (-) ^a
WHemlock trees 20-40cm dbh ^x	0.6 (1.1) ^a	0.0 (0.2) ^{ac}	0.9 (1.4) ^{ad}	3.7 (4.6) ^b
WRCedar trees 20-40cm dbh ^x	0.0 (-) ^a	0.1 (0.3) ^a	0.8 (1.1) ^b	0.3 (0.6) ^{ab}
Douglas-fir 20-40cm dbh	0.6 (1.1)	0.9 (2.7)	0.8 (1.4)	0.7 (0.6)
Trees 20-40cm dbh	7.7 (4.5)	8.8 (4.5)	10.8 (4.8)	6.3 (4.0)
Conifer trees 20-40cm dbh	1.2 (1.4) ^{ab}	1.0 (2.6) ^a	2.5 (2.4) ^b	4.7 (4.6) ^{ab}
Deciduous trees 20-40cm dbh ^x	6.5 (4.3) ^{ab}	7.8 (4.5) ^{ab}	8.3 (4.5) ^a	1.7 (0.6) ^b
Shade tolerant 20-40cm dbh ^x	0.6 (1.1) ^a	0.1 (0.3) ^a	1.7 (2.1) ^b	4.0 (5.2) ^b
Shade intolerant 20-40cm dbh	7.0 (4.6)	8.5 (4.7)	8.7 (4.7)	2.3 (1.2)
<u>Trees 40-60cm dbh</u>				
BL maple trees 40-60cm dbh ^x	2.4 (1.8) ^a	1.8 (1.5) ^a	1.7 (1.5) ^{ab}	0.0 (-) ^b
RAlder trees 40-60cm dbh	0.5 (0.8)	0.8 (0.9)	0.5 (0.8)	0.0 (-)
WHemlock trees 40-60cm dbh	0.0 (-)	0.0 (0.2)	0.4 (0.9)	0.3 (0.6)
WRCedar trees 40-60cm dbh ^x	0.0 (-) ^a	0.0 (0.2) ^a	0.3 (0.6) ^b	1.0 (1.0) ^b
Douglas-fir 40-60cm dbh	0.5 (0.8)	0.7 (1.1)	0.9 (1.3)	0.3 (0.6)
Trees all species 40-60cm dbh	3.4 (1.6)	3.5 (2.1)	4.0 (1.6)	1.7 (1.2)
Conifer trees 40-60cm dbh ^x	0.5 (0.8) ^a	0.9 (1.2) ^{ab}	1.6 (1.4) ^b	1.7 (1.2) ^{ab}
Deciduous trees 40-60cm dbh ^x	3.0 (1.8) ^a	2.6 (1.7) ^a	2.5 (1.8) ^a	0.0 (-) ^b
Shade tolerant 40-60cm dbh ^x	0.0 (-) ^a	0.1 (0.3) ^a	0.7 (1.2) ^b	1.3 (0.6) ^b
Shade intolerant 40-60cm dbh ^x	3.4 (1.6) ^a	3.3 (1.9) ^a	3.2 (1.3) ^a	0.3 (0.6) ^b
<u>Trees >60cm dbh</u>				
BL maple trees >60cm dbh	0.5 (0.8)	0.5 (0.7)	0.3 (0.5)	0.0 (-)
WHemlock trees >60cm dbh	0.0 (0.2)	0.0 (0.2)	0.3 (0.4)	0.3 (0.6)
Douglas-fir >60cm dbh ^x	1.1 (1.4) ^a	0.8 (1.5) ^a	0.5 (0.7) ^a	5.0 (1.7) ^b
Trees all species >60cm dbh ^x	1.7 (1.5) ^a	1.4 (1.4) ^a	1.3 (1.3) ^a	5.3 (1.2) ^b
Conifer trees >60 cm dbh ^x	1.1 (1.4) ^a	0.8 (1.5) ^a	0.9 (1.1) ^a	5.3 (2.1) ^b
Deciduous trees >60cm dbh	0.5 (0.8)	0.6 (0.7)	0.3 (0.6)	0.0 (-)
Shade tolerant >60cm dbh ^x	0.0 (0.2) ^a	0.0 (0.2) ^a	0.5 (0.8) ^b	0.3 (0.6) ^{ab}
Shade intolerant >60cm dbh ^x	1.6 (1.4) ^{ab}	1.4 (1.4) ^a	0.8 (0.8) ^a	5.0 (1.7) ^b
n	24	24	24	3

Tree species that were present in less than two sections were considered rare and omitted.

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression. A superscript of "x" denotes that the square root of the variable was used in statistical analyses.

APPENDIX G

MEAN INCREMENT GROWTH RATE (CM/YEAR) BY DECADE FOR THE
PREVIOUS TEN DECADES BY TREE SPECIES

MEAN INCREMENT GROWTH RATE (CM/YEAR) BY DECADE FOR THE
PREVIOUS TEN DECADES BY TREE SPECIES

FIRST DECADE (0-10 Years Ago)

Tree Species	City	Mid	Far	OG
Bigleaf Maple n	1.63 (0.53) 95	1.71 (0.56) 93	1.63 (0.56) 79	1.50 (0.69) 5
Red Alder n	1.65 (0.61) 33	1.69 (0.51) 57	1.54 (0.41) 38	0.0 (-) 0
Cherry n	0.0 (-) 0	1.68 (0.19) 5	1.27 (0.40) 7	0.0 (-) 0
Dogwood n	1.43 (0.43) 4	1.43 (0.39) 2	0.0 (-) 0	1.60 (-) 1
Douglas-fir n	1.30 (0.62) 43	1.32 (0.48) 40	1.33 (0.43) 40	1.00 (0.41) 12
Western Hemlock n	1.22 (0.59) 21	1.42 (0.56) 5	1.27 (0.53) 32	1.05 (0.29) 11
Western Red cedar n	0.0 (-) 0	1.07 (0.33) 7	1.22 (0.43) 40	0.96 (0.22) 8
Grand fir n	1.22 (0.34) 15	1.37 (0.46) 9	1.24 (0.91) 7	0.0 (-) 0
All trees n	1.48 (0.58) ^a 214	1.59 (0.54) ^{ac} 223	1.43 (0.52) ^{ad} 247	1.09 (0.42) ^b 37

SECOND DECADE (10-20 Years Ago)

Tree Species	City	Mid	Far	OG
Bigleaf Maple ^x n	2.24 (0.69) ^a 94	2.55 (0.77) ^b 90	2.53 (0.88) ^{ab} 79	1.15 (0.40) ^c 5
Red Alder n	2.17 (0.97) 33	2.37 (0.57) 57	2.40 (0.67) 38	0.0 (-) 0
Douglas-fir ^x n	1.57 (0.68) ^a 43	1.72 (0.59) ^a 40	1.83 (0.71) ^a 40	0.96 (0.37) ^b 12
Western Hemlock n	1.46 (0.71) 21	1.72 (0.59) 5	1.63 (0.78) 32	1.17 (0.49) 11
Western Red cedar ^x n	0.0 (-) ^{ab} 0	0.81 (0.31) ^a 7	1.50 (0.72) ^b 40	1.22 (0.51) ^{ab} 8
Grand fir n	1.71 (0.57) 14	1.69 (0.64) 9	2.26 (2.04) 7	0.0 (-) 0
All trees ^x n	1.95 (0.80) ^a 212	2.20 (0.80) ^b 219	2.09 (0.92) ^{ab} 247	1.13 (0.45) ^c 37

THIRD DECADE (20-30 Years Ago)

Tree Species	City	Mid	Far	OG
Bigleaf Maple ^x n	2.56 (1.03) ^a 90	2.95 (0.74) ^b 78	2.87 (1.06) ^{ab} 70	1.26 (0.32) ^c 5
Red Alder n	2.44 (1.17) 33	2.74 (0.71) 50	2.73 (0.83) 38	0.0 (-) 0
Douglas-fir ^x n	1.74 (0.85) ^a 43	1.69 (0.57) ^a 34	2.09 (0.99) ^a 33	0.99 (0.45) ^b 12
Western Hemlock n	1.62 (1.07) ^{ab} 19	1.73 (0.68) ^{ab} 4	1.99 (1.07) ^a 27	1.13 (0.60) ^b 11
Western Red cedar ^x n	0.0 (-) ^{ab} 0	0.77 (0.36) ^a 6	1.68 (0.71) ^b 35	1.41 (0.52) ^{ab} 8
Grand fir n	1.00 (0.60) 9	1.82 (1.48) 8	1.98 (1.09) 6	0.0 (-) 0
All trees ^x n	2.20 (1.10) ^a 196	2.48 (0.96) ^b 189	2.38 (1.05) ^{ab} 248	1.18 (0.51) ^c 37

FOURTH DECADE (30-40 Years Ago)

Tree Species	City	Mid	Far	OG
Bigleaf Maple ^x n	2.54 (1.12) ^a 83	3.01 (0.66) ^b 59	2.87 (0.95) ^{ab} 49	1.35 (0.17) ^c 4
Red Alder n	2.78 (1.15) 32	2.85 (0.64) 45	3.32 (1.22) 33	0.0 (-) 0
Douglas-fir ^x n	2.11 (1.32) ^a 42	1.83 (0.66) ^a 30	1.82 (0.68) ^a 28	1.03 (0.49) ^b 12
Western Hemlock ^x n	1.44 (0.79) ^{ab} 17	1.58 (0.41) ^{ab} 4	2.17 (1.26) ^a 22	1.01 (0.39) ^b 9
Western Red cedar ^x n	0.0 (-) ^{ab} 0	1.21 (0.28) ^{ab} 3	1.66 (0.50) ^a 23	1.14 (0.26) ^b 8
Grand fir n	0.91 (0.37) 6	1.50 (0.90) 4	2.05 (1.57) 3	0.0 (-) 0
All trees ^x n	2.33 (1.21) ^a 181	2.58 (0.86) ^b 148	2.47 (1.13) ^{ab} 167	1.09 (0.38) ^c 34

FIFTH DECADE (40-50 Years Ago)

Tree Species	City	Mid	Far	OG
Bigleaf Maple ^x n	2.35 (0.78) ^a 61	3.01 (0.72) ^b 40	2.92 (1.01) ^{ab} 26	1.61 (0.16) ^c 4
Red Alder n	2.88 (1.42) 25	2.78 (0.55) 26	3.32 (1.21) 19	0.0 (-) 0
Douglas-fir ^x n	1.99 (0.87) ^a 35	1.95 (0.50) ^a 27	1.81 (0.65) ^a 25	1.01 (0.34) ^b 12
Western Hemlock ^x n	1.35 (0.57) ^a 15	1.54 (0.54) ^{ab} 2	2.09 (0.68) ^b 15	1.18 (0.62) ^a 9
Western Red cedar ^x n	0.0 (-) ^{ab} 0	1.15 (0.21) ^{ab} 2	2.09 (1.19) ^a 18	1.15 (0.35) ^b 7
All trees ^x n	2.22 (1.03) ^a 139	2.54 (0.81) ^b 103	2.44 (1.11) ^{ab} 111	1.17 (0.44) ^c 33

SIXTH DECADE (50-60 Years Ago)

Tree Species	City	Mid	Far	OG
Bigleaf Maple ^x n	2.36 (0.72) ^a 41	2.95 (0.56) ^b 23	2.68 (0.81) ^{ab} 17	1.33 (0.21) ^c 3
Red Alder ^x n	2.10 (0.75) ^a 15	2.78 (0.89) ^b 16	2.74 (0.62) 11	0.0 (-) 0
Douglas-fir ^x n	1.81 (0.58) ^a 32	1.95 (0.49) ^a 24	1.93 (0.55) ^a 23	1.20 (0.42) ^b 12
Western Hemlock n	1.64 (0.55) 13	1.97 (0.30) 2	2.07 (0.68) 10	1.72 (0.46) 5
Western Red cedar ^x n	0.0 (-) 0	1.64 (0.10) 2	2.12 (0.63) ^a 13	1.11 (0.40) ^b 4
All trees ^x n	2.05 (0.71) ^a 102	2.46 (0.79) ^b 69	2.29 (0.78) ^{ab} 80	1.31 (0.44) ^c 24

SEVENTH DECADE (60-70 Years Ago)

Tree Species	City	Mid	Far	OG
Bigleaf Maple n	2.18 (0.70) 24	2.61 (0.45) 17	2.46 (0.43) 6	1.55 (0.35) 2
Red Alder n	1.79 (1.00) 11	2.60 (0.43) 5	2.74 (0.37) 5	0.0 (-) 0
Douglas-fir ^x n	1.74 (0.66) ^a 30	1.99 (0.43) ^a 21	1.93 (0.55) ^a 22	1.26 (0.59) ^b 12
Western Hemlock n	1.56 (0.59) 7	1.60 (-) 1	1.98 (0.35) 7	1.49 (0.30) 4
Western Red cedar ^x n	0.0 (-) ^{ab} 0	1.50 (-) ^{ab} 1	1.98 (0.26) ^a 8	1.06 (0.28) ^b 4
All trees ^x n	1.88 (0.75) ^a 72	2.22 (0.60) ^b 47	2.16 (0.57) ^b 51	1.29 (0.49) ^c 22

Tree species that were present in less than two sections or for which the total n was less than 12 were considered infrequent or rare species and were omitted.

Results of Bonnferroni/Dunn testing is shown for each variable as superscripts for means that are significantly different from one another. For a given variable, means with different superscripts are significantly different from one another (e.g., for any given dependent variable, all means with a superscript of a are significantly different from those with a superscript of b). Significance was determined after the variables were transformed in the same manner as used in multiple regression testing.

A superscript of "x" denotes that the square root of the variable was used; a superscript of "*" denotes that the natural log of the variable + 1 was used.

APPENDIX H

GUILD STATUS AND MIGRATORY STATUS OF ALL BIRDS OBSERVED IN FOREST PARK AND THE OLD-GROWTH STAND

TABLE OF BIRD SPECIES BY GUILD STATUS. THE MIGRATORY STATUS
OF EACH BIRD IS LISTED

<u>Guild Status / Common Name</u>	<u>Migratory Status</u>
Predatory birds:	
Sharp-shinned hawk	S
Cooper's hawk	S
Red-tailed hawk	R
Osprey	S
Bald eagle	R
Northern pygmy-owl	R
Western screech-owl	R
Ground gleaners:	
Ruffed grouse	R
California quail	R
Rock dove	R
Northern flicker	R
Scrub jay	R
Steller's jay	R
American crow	R
Common raven	R
Bewick's wren	R
House wren	S
Winter wren	R
American robin	S
Varied thrush	R
Hermit thrush	S
House sparrow	R
European starling	R
Brown-headed cowbird	S
Evening grosbeak	R
Purple finch	R
House finch	R
Spotted towhee	R
Dark-eyed junco	R
Song sparrow	S
Fox sparrow	S
White-crowned sparrow	S

<u>Guild Status / Common Name</u>	<u>Migratory Status</u>
Foliage gleaners:	
Band-tailed pigeon	S
Mourning dove	S
Black-capped chickadee	R
Chestnut-backed chickadee	R
Bushtit	R
Swainson's thrush	N
Golden-crowned kinglet	R
Ruby-crowned kinglet	R
Cedar waxwing	S
Solitary vireo	N
Warbling vireo	N
Hutton's vireo	R
Orange-crowned warbler	N
Nashville warbler	N
Yellow-rumped warbler	N
Hermit warbler	N
Townsend's warbler	N
Black-throated gray warbler	N
McGillivray's warbler	N
Yellow warbler	N
Wilson's warbler	N
Western tanager	N
Black-headed grosbeak	N
Pine siskin	R
Bark gleaners:	
Hairy woodpecker	R
Downy woodpecker	R
Pileated woodpecker	R
Red-breasted nuthatch	R
White-breasted nuthatch	R
Brown creeper	R
Aerial gleaners:	
Rufous hummingbird	S
Olive-sided flycatcher	N
Pacific-slope flycatcher	N
Western wood-peewee	N
Violet-green swallow	N

Guild and migratory status as per Ehrlich et al. 1988

APPENDIX I

COMMON AND SCIENTIFIC NAMES (AUTHORITIES) OF BIRD SPECIES
COUNTED AT SITES, AND ABBREVIATIONS USED IN TEXT AND TABLES

COMMON AND SCIENTIFIC NAMES (AUTHORITIES) OF BIRD SPECIES
COUNTED AT SITES, AND ABBREVIATIONS USED IN TEXT AND TABLES

Common Name	Scientific Name (Authority)	Abbreviation
Canada goose	<i>Branta canadensis</i> (Linnaeus)	CAGO
Sharp-shinned hawk	<i>Accipiter striatus</i> (Vieillot)	SSHA
Cooper's hawk	<i>Accipiter cooperii</i> (Vieillot)	COHA
Red-tailed hawk	<i>Buteo jamaicensis</i> (Gmelin)	RTHA
Osprey	<i>Pandion haliaetus</i> (Linnaeus)	OSPR
Bald eagle	<i>Haliaeetus leucocephalus</i> (Linnaeus)	BAEA
Great blue heron	<i>Ardea herodias</i> (Linnaeus)	GBHE
Ruffed grouse	<i>Bonasa umbellus</i> (Linnaeus)	RUGR
California quail	<i>Callipepla californica</i> (Shaw)	CAQU
Rock dove	<i>Columba livia</i> (Gmelin)	RODO
Band-tailed pigeon	<i>Columba fasciata</i> (Say)	BTPI
Mourning dove	<i>Zenaida macroura</i> (Linnaeus)	MODO
Great-horned owl	<i>Bubo virginianus</i> (Gmelin)	GHOW
Northern pygmy-owl	<i>Glaucidium gnoma</i> (Wagler)	NPOW
Western screech-owl	<i>Otus kennicottii</i> (Elliot)	WSOW
Rufous hummingbird	<i>Selasphorus rufus</i> (Gmelin)	RUHU
Northern flicker	<i>Colaptes auratus</i> (Linnaeus)	NOFL
Hairy woodpecker	<i>Picoides villosus</i> (Linnaeus)	HAWO
Downy woodpecker	<i>Picoides pubescens</i> (Linnaeus)	DOWO
Pileated woodpecker	<i>Dryocopus pileatus</i> (Linnaeus)	PIWO
Olive-sided flycatcher	<i>Contopus borealis</i> (Swainson)	OSFL
Pacific-slope flycatcher	<i>Empidonax difficilis</i> (Baird)	PSFL
Western wood-pewee	<i>Contopus sordidulus</i> (Sclater)	WWPE
Violet-green swallow	<i>Tachycineta thalassina</i> (Swainson)	VGSW
Scrub jay	<i>Aphelocoma coerulescens</i> (Bosc)	SCJA
Steller's jay	<i>Cyanocitta stelleri</i> (Gmelin)	STJA
American crow	<i>Corvus brachyrhynchos</i> (Brehm)	AMCR
Common raven	<i>Corvus corax</i> (Linnaeus)	CORA

Common Name	Scientific Name (Authority)	Abbreviation
Black-capped chickadee	<i>Parus atricapillus</i> (Linnaeus)	BCCH
Chestnut-backed chickadee	<i>Parus rufescens</i> (Townsend)	CBCH
Bushtit	<i>Psaltiriparus minimus</i> (Townsend)	BUSH
Red-breasted nuthatch	<i>Sitta canadensis</i> (Linnaeus)	RBNU
White-breasted nuthatch	<i>Sitta carolinensis</i> (Latham)	WBNU
Brown creeper	<i>Certhia familiaris</i> (Bonaparte)	BRCR
Bewick's wren	<i>Thryomanes bewickii</i> (Audubon)	BEWR
House wren	<i>Troglodytes aedon</i> (Vieillot)	HOWR
Winter wren	<i>Troglodytes troglodytes</i> (Linnaeus)	WTWR
American robin	<i>Turdus migratorius</i> (Linnaeus)	AMRO
Varied thrush	<i>Ixoreus naevius</i> (Gmelin)	VATH
Swainson's thrush	<i>Catharus ustulata</i> (Nuttall)	SWTH
Hermit thrush	<i>Hylocichla guttata</i> (Pallas)	HETH
Golden-crowned kinglet	<i>Regulus satropa</i> (Lichtenstein)	GCKI
Ruby-crowned kinglet	<i>Regulus calendula</i> (Linnaeus)	RCKI
Cedar waxwing	<i>Bombycilla cedrorum</i> (Vieillot)	CEWA
European starling	<i>Sturnus vulgaris</i> (Linnaeus)	EUST
Solitary vireo	<i>Vireo solitarius</i> (Wilson)	SOVI
Warbling vireo	<i>Vireo gilvus</i> (Vieillot)	WAVI
Hutton's vireo	<i>Vireo huttoni</i> (Cassin)	HUVI
Orange-crowned warbler	<i>Vermivora celata</i> (Say)	OCWA
Nashville warbler	<i>Vermivora ruficapilla</i> (Wilson)	NAWA
Yellow-rumped warbler	<i>Dendroica coronata</i> (Linnaeus)	YRWA
Hermit warbler	<i>Dendroica occidentalis</i> (Townsend)	HEWA
Townsend's warbler	<i>Dendroica townsendi</i> (Townsend)	TOWA
Black-throated gray warbler	<i>Dendroica nigrescens</i> (Townsend)	BTGW
McGillivray's warbler	<i>Oporornis tolmiei</i> (Townsend)	MCWA
Yellow warbler	<i>Dendroica petechia</i> (Linnaeus)	YEWA
Wilson's warbler	<i>Wilsonia pusilla</i> (Wilson)	WTWA
House sparrow	<i>Passer domesticus</i> (Linnaeus)	HOSP
Brown-headed cowbird	<i>Molothrus ater</i> (Boddaert)	BHCO

<u>Common Name</u>	<u>Scientific Name (Authority)</u>	<u>Abbreviation</u>
Western tanager	<i>Piranga ludoviciana</i> (Wilson)	WETA
Black-headed grosbeak	<i>Pheucticus melanocephalus</i> (Swainson)	BHGR
Evening grosbeak	<i>Coccothraustes vespertinus</i> (Cooper)	EVGR
Purple finch	<i>Carpodacus purpureus</i> (Gmelin)	PUFI
House finch	<i>Carpodacus mexicanus</i> (Müller)	HOFI
Pine siskin	<i>Carduelis pinus</i> (Wilson)	PISI
Spotted towhee	<i>Pipilo erythrophthalmus</i> (Linnaeus)	SPTO
Dark-eyed junco	<i>Junco hyemalis</i> (Linnaeus)	DEJU
Song sparrow	<i>Melospiza melodia</i> (Wilson)	SOSP
Fox sparrow	<i>Passerella iliaca</i> (Merrem)	FOSP
White-crowned sparrow	<i>Zonotrichia leucophrys</i> (Forster)	WCSP

All scientific names and authorities via Ehrlich et al 1988. Common names conform to those adopted by the Oregon Bird Records Committee 1998.

APPENDIX J

REGRESSION COEFFICIENTS FOR BIRD SPECIES ABUNDANCE,
RICHNESS, AND DIVERSITY

REGRESSION COEFFICIENTS FOR BIRD SPECIES ABUNDANCE, RICHNESS, AND DIVERSITY

Bird Species Abundance, Richness, and Diversity -- Breeding Season

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt (# birds)	7.89	-0.04	-0.18	2.04	1.70	-0.84	0.37	-0.04	-0.21
sqrt (# urban birds)	5.17	-0.04	-1.05	-0.94	-1.04	-0.59	-1.07	0.07	-0.99
sqrt (# urban species)	1.17	-0.01	-0.20	0.36	-0.29	0.04	-0.43	0.05	-0.78
sqrt (# edge birds)	5.64	-0.16	-0.46	-0.81	-0.09	-0.57	0.10	-0.08	-0.46
sqrt (# forest birds)	2.06	0.10	0.49	1.18	1.05	0.01	-0.05	0.04	-0.03
sqrt (# resident birds)	5.40	-0.09	0.33	2.67	2.87	-1.48	0.10	-0.07	-0.20
sqrt (# short distance)	7.10	-0.06	-1.20	-1.40	-1.20	-0.64	-0.57	0.03	-0.79
sqrt (# predatory birds)	-0.81	0.02	-0.41	0.87	1.22	-0.19	1.23	-0.13	0.39
sqrt (# ground gleaners)	7.63	-0.06	-0.86	-0.20	0.91	-1.21	-0.69	-0.02	-0.58
sqrt (# bark gleaners)	1.42	0.03	-0.31	1.32	2.48	-0.40	-0.26	0.05	0.35
sqrt (# hawks)	-0.57	0.02	-0.33	0.89	0.50	-0.32	1.25	-0.19	0.41
sqrt (# STJA)	-2.44	0.03	0.31	2.80	-0.93	0.28	2.11	-0.08	-0.59
sqrt (# BRGR)	-0.11	0.00	0.52	1.72	0.89	-0.43	1.14	-0.05	-0.09
sqrt (# WTWR)	3.16	0.05	-0.57	-0.40	1.81	-0.45	-0.87	0.07	0.39
sqrt (# SWTH)	-0.20	0.10	-0.86	1.24	-0.64	0.51	0.91	-0.07	-0.47
sqrt (# HOFI)	1.33	-0.04	-0.23	-0.02	-0.42	-0.22	-1.08	0.12	-0.24
sqrt (# RSTO)	3.62	-0.12	1.57	-0.76	0.82	-1.19	0.24	0.05	-0.63
sqrt (# SOSPI)	8.61	-0.45	1.31	0.11	-1.91	-2.19	0.92	-0.33	-1.89

Bird Species Abundance, Richness, and Diversity -- Winter

Dependent Variable	Intercept	PDX	ROAD	NWI	EDGE	FOREST	CENTER	HOUSE	OG
sqrt (total # birds)	1.58	-0.01	-0.27	-0.61	1.80	0.87	1.83	0.08	7.21
sqrt (# resident birds)	1.35	-0.01	-0.30	-0.39	1.71	0.87	1.96	0.06	6.84
SQRT (# short d migrants)	0.09	0.02	-0.30	0.30	-1.19	0.06	0.29	0.04	-0.68
sqrt (# foliage gleaners)	2.58	-0.08	-0.32	-1.59	1.87	0.59	1.09	0.01	8.12
sqrt (# bark gleaners)	-1.38	0.04	-0.25	1.14	0.53	0.42	1.82	-0.01	1.09
sqrt (# BRGR)	-1.01	0.00	0.18	0.86	0.80	0.22	1.17	0.02	1.24
sqrt (# GCKI)	0.61	-0.03	-0.26	-0.82	1.26	0.94	0.80	0.08	6.30

APPENDIX K

REGRESSION COEFFICIENTS FOR PLANT COMMUNITY
CHARACTERISTICS AS APPLIED TO THE AVIAN COMMUNITY

REGRESSION COEFFICIENTS FOR PLANT COMMUNITY CHARACTERISTICS

Avian Community Parameters -- Breeding Season Birds

Dependent Variable	Intercept	TREE	MAPLE	CEDAR	SHRUB	THIMB	H'H	EXOTIC	FHD
sqrt (# birds)	11.31	0.00	-0.03	-0.02	0.34	-0.05	-1.24	0.46	0.04
sqrt (# urban birds)	-0.29	0.15	-0.01	0.04	0.06	0.07	1.06	1.36	-0.14
sqrt (# edge birds)	1.39	-0.22	0.02	-0.11	0.00	0.00	1.57	0.34	-0.04
sqrt (# edge species)	1.39	-0.03	0.01	-0.02	0.02	0.01	0.37	0.30	-0.05
sqrt (# short distance)	2.05	0.05	-0.01	0.05	0.06	0.09	1.26	1.38	-0.19
sqrt (# bark gleaners)	4.87	-0.02	-0.03	0.11	0.04	-0.04	-0.56	0.20	-0.03
sqrt (# raptors)	3.86	-0.01	-0.01	-0.08	0.22	0.02	-1.85	-0.21	0.04
sqrt (# PIWO)	1.25	-0.14	-0.01	-0.08	-0.34	-0.10	0.19	0.22	0.17
sqrt (# STJA)	4.61	0.12	-0.02	-0.09	0.13	0.01	-2.14	0.91	0.03
sqrt (# AMRO)	0.80	0.12	-0.01	0.07	-0.02	0.09	0.85	1.14	-0.15
sqrt (# SWTH)	5.85	-0.20	0.00	0.01	0.11	-0.01	-1.75	-0.52	0.13
sqrt (# WETA)	2.51	0.10	-0.02	0.25	0.10	-0.04	-0.49	0.29	-0.09
sqrt (# HOFI)	-3.24	0.19	-0.01	-0.06	0.19	-0.02	0.80	0.49	-0.03
sqrt (# RSTO)	1.60	-0.03	0.00	-0.37	-0.08	0.00	0.54	0.14	-0.01
sqrt (# SOSP)	0.59	-0.31	0.02	-0.04	0.13	0.00	1.08	0.07	0.00

Avian Community Parameters -- Winter Birds

Dependent Variable	Intercept	TREE	MAPLE	CEDAR	SHRUB	THIMB	H'H	EXOTIC	FHD
sqrt (# birds)	9.73	0.19	-0.06	0.17	0.89	0.03	-3.40	0.04	-0.07
sqrt (# urban birds)	-1.09	0.10	-0.01	-0.07	0.02	0.00	0.26	0.03	-0.01
sqrt (# forest birds)	4.04	-0.03	-0.02	-0.11	0.21	-0.04	-1.49	0.54	0.06
sqrt (# residents)	9.68	0.16	-0.06	0.17	0.83	0.03	-3.30	-0.20	-0.06
sqrt (# foliage gleaners)	7.28	0.20	-0.05	0.27	1.02	0.07	-2.99	-0.74	-0.09
sqrt (# bark gleaners)	5.09	0.06	-0.03	-0.13	0.23	-0.02	-1.92	0.54	0.03
sqrt (# BCCB)	3.79	0.15	-0.02	0.23	0.73	0.05	-1.61	-0.19	-0.12
sqrt (# BRGR)	3.79	0.03	-0.02	-0.08	0.26	-0.03	-1.50	0.52	0.02
sqrt (# GCKI)	6.13	0.11	-0.05	0.21	0.68	0.04	-2.52	-0.82	0.00