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"New Forestry" in Practice: A Survey of Mortality in Green Tree Retention Harvest Units, Western Cascades, Oregon

by

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Abstract

The ecological integrity of the Pacific Northwest's forests depends not only on a system of reserves, but also on changing traditional silvicultural practices in managed stands to reflect environmental as well as commodity values. Biological and functional diversity depend on the great structural complexity unique to the region's late-successional forests. Green tree retention is one experimental strategy used to maintain structural complexity after harvest. However, the mortality of trees retained after harvest is unknown. Fourty-four cutting units harvested 1 to 10 years ago in the Western Cascades, Oregon, were surveyed for windthrow and other mortality. Total windthrow ranged from 0 to 58%, and averaged 15.6% of retained green trees. Other mortality--trees that died standing--averaged 15.9%. Average annual windthrow, which takes into account the number of storm seasons since harvest, averaged 4.7%, and ranged from 0 to 22.6%. Sites harvested within three years before the severe storms of 1990 had the highest average annual rates of windthrow. Time of harvest was the only factor that explained a marginally significant amount of between-site variation in windthrow, indicating the importance of the timing of large storms. This result, in addition to the complex interactions between wind direction, topography, and numerous site factors, may prohibit accurate prediction of risk. A simulation assuming the retained trees become windfirm after 5 years showed that from 25 to 75% total windthrow can be expected after 100 years. The potential for severe windthrow exists in virtually every site. To accommodate this damage, more green trees must be retained, especially when creation of nesting habitat is the primary objective.

The fate of old-growth forests in the Pacific Northwest has become one of the defining issues in the national debate on natural resource management. Groups as divergent as environmentalists and the Wise Use movement have adopted the northern spotted owl, and the forests it represents, as a symbol of their cause. President Clinton's Forest Summit in April, 1993, recognized the importance and difficulty of the issue, and began the latest chapter in a long series of planning projects.

But while the attention and controversy has focused on the creation of old-growth reserves, the ecological integrity of the regional forest also depends on revamping existing practices in managed areas. The Forest Ecosystem Management Assessment Team (FEMAT), appointed by Clinton, writes, "Stand level practices that have created dense young plantations...have altered the typical pathways by which stands develop into old-growth" (FEMAT, 1993: IV-76). Unless logged forests are managed to provide additional habitat for old-growth restricted species and link the network of reserves, "the current and future late-successional ecosystem is at a relatively high risk of loss or inadequate development" (FEMAT, 1993: p. IV-76).

Green tree retention is one of the new silvicultural methods designed to improve the ecological function of managed forests. Snags and large live trees retained in the cutting unit supply wildlife habitat and may accelerate a return to old-growth conditions. But green tree retention has been applied only within the last seven years and remains experimental (Franklin, 1992). The survival of retained trees after harvest is unknown. Since retained trees are exposed to stronger winds in the open cutting unit than under the protection of an intact canopy, windthrow is of special concern. Implicit in these questions is the possibility that high windthrow, and perhaps other forms of mortality, could compromise the objectives of green tree retention. The purpose of this study is to describe and analyze patterns of windthrow and other mortality in recently harvested green tree retention cutting units, and to consider the implications of these results.

Pacific Northwest forests and "New Forestry"

The forests of the Pacific Northwest represent one of the only mesic temperate forests dominated by conifers. At similar latitudes elsewhere in the northern hemisphere, hardwoods, or a hardwood-conifer mix, prevail, with conifers limited to resource poor sites. In the Northwest, a combination of wet, moderate winters and summer drought allows conifers to out-compete trees whose productivity is limited by season and moisture (Waring and Franklin, 1979). This unique climactic regime also creates conditions for exceptional tree size and longevity: "Every single coniferous genus represented finds its largest (and often longest lived) specific representative here--and often its second and third largest as well" (Franklin and Dyrness, 1973: p. 54). Douglas-fir, the dominant tree species in much of the region, can grow to 90 meters and live over 1000 years. A stand of these giants accumulates more biomass than any temperate plant community and, possibly, any community world-wide (Franklin and Dyrness, 1973). The individual trees, the stands they form, and the greater ecosystem have no equivalent.

These structural elements--large live trees, snags, and downed logs--more than any other characteristics, form the basis of old-growth forests. Large living trees are the source of snags and large downed logs, and the gaps created by their death allow smaller shade tolerant trees to grow. All these structures are essential pieces of an old-growth forest. Late successional forests are *always* structurally distinct from younger forests (Franklin and Spies, 1991). This mix of structural attributes provides habitat for a wide range of organisms and performs ecosystem functions less developed in younger stands.

Old-growth's mix of large and small trees, snags, and coarse woody debris (CWD), creates structural complexity and spatial heterogeneity that in turn supports a higher level of biodiversity than younger closed-canopy forests (Franklin, 1992; Hansen et al., 1991). High variation of characteristics such as tree density create a wide range of niches in a small space (Swanson and Franklin, 1992). The spotted-owl, marbled-murrelet and other cavity nesters have drawn attention to the importance of large trees and snags.

Less publicized are the invertebrates, fungi, and microbes that are also more abundant in late-successional forests than in younger stands (Perry et al., 1989; Franklin, 1992). Insects and foliose lichens utilize the uneven canopy and large leaf area of old-growth trees (Franklin, 1992). Large rotting logs support another suite of species and, underground, mycorrhizal fungi form important symbiotic relationships with trees. Insect and pest predators are more common in older forests (Schowalter, 1989). Aquatic species depend on forest structure also. The canopy shades streams, keeping water temperature low, and CWD creates essential pools and spawning habitat (Harmon et al., 1986; Franklin et al., 1981; Swanson and Franklin, 1992). While 9 species are known to depend on old-growth forests for survival, many species prefer old-growth to other forests conditions (Hansen et al., 1991).

Structural complexity also contributes to the ecosystem functions performed by old-growth forests. CWD acts as a dam for sediment on hillsides and stabilizes stream channels, limiting erosion and maintaining water quality (Swanson and Dyrness, 1975; Franklin and Spies, 1991b). The high leaf areas of old-growth forests, 9 to 15 meters^2 per square meter of ground surface, compared to 6 to 8 meters^2 in young stands, intercept precipitation and mitigate flooding (Franklin, 1992; Swanson, Franklin and Sedell, 1990). This function is especially important in warm rain-on-snow events, when open areas can produce significantly more water, increasing peak stream flows. At other times, high leaf area can increase effective moisture by fog-drip (Harr, 1982). In addition to limiting erosion, old-growth forests in the Pacific Northwest lose fewer nutrients than younger stands (Franklin and Spies, 1991b; Franklin et al., 1981). One reason for this is that nitrogen fixing often depends on the presence of structural elements associated with old-growth, or on interactions that take many years to develop: Nitrogen fixing lichens, most notably *Lobaria oregana*, grow in the crowns of large trees; rotting wood is a site of N-fixing, as is the rhizosphere, which features the symbiosis between N-fixing mycorrhizal fungi and many tree species (Franklin, 1992). Ten to forty percent of total photosynthate goes to the rhizosphere, evidence of the importance of below-ground processes (Perry et al., 1989).

Many researchers believe that the biological diversity and functional diversity of old-growth forests creates ecological stability. Plant and soil organism mutualism creates the conditions which "allow the systems to persist," so that severing these links leads to degradation (Perry et al., 1989). Another model suggests that stability, defined as the persistence of community composition, depends on individual species and not emergent properties: The more original species in the community, the less chance of invaders succeeding after disturbance (Halpern, 1988). Long-term site productivity, another type of stability, depends on the presence of CWD because of its importance in the nutrient cycle (Harmon et al., 1986). At the landscape level, the limitation of erosion and nutrient loss by old-growth forests provides resistance against catastrophic storm events and long-term degradation of soil and water (Franklin, 1993).

The concept of "biological legacies" explains how forests maintain biological and functional diversity after natural disturbances. Early studies of fire history in the Cascades tended to treat fire, the primary disturbance agent in the region, as a catastrophic, stand replacing event (Hemstrom, 1982; Hemstrom and Franklin, 1982). More recent studies have focused on fire as a partial disturbance. Fires typically leave stands with a high degree of structural complexity created by remnant live trees, snags, and CWD (Franklin, 1992; Hansen et al., 1991). Since trees do survive fires, regenerating stands often have two or more age-classes with extremely variable tree densities (Swanson, Franklin, and Sedell, 1990). The variability in fire severity produces a mosaic of small patches (Morrison and Swanson, 1990). Fire severity also depends on a site's position along the gradient from infrequent, catastrophic fires in the north to frequent, low intensity fires in the south. While some sites may burn every 20 years, return intervals in other locations may be greater than 400 years (Morrison and Swanson, 1990); one site in Mt. Rainier National Park has not burned for over 1,200 years (Hemstrom and Franklin, 1982). The degree of biological legacy left by a fire depends on local patchiness and the regional gradient, both of which are characterized by great variability.

Other disturbances leave biological legacies as well. Windthrow creates large quantities of CWD, though it does not favor reestablishment by early seral species such as Douglas-fir, as fire does (Morrison and Swanson, 1990). Windthrow may accelerate succession by releasing shade tolerant, understory trees. Pathogens have a similar effect, but leave behind standing snags. As a result of these legacies, young natural stands often have very high levels of structural complexity, especially CWD. Levels of CWD fall in mature stands, where decay outstrips production, before rising again in the old-growth stage (Hansen, et al. 1991; Spies and Franklin, 1991).

Traditional timber harvesting, in contrast, leaves none of the biological legacies that allow individual species and ecosystem processes to persist after natural disturbances. Clear-cutting and reforestation reduce a complex forest to an even-aged stand with no snags and little CWD. Snag and log abundance typically falls to 6% of natural levels (Hansen et al., 1991). Organisms that depend on these structures for habitat are displaced, including aquatic species that depend on the canopy for shading and CWD for habitat structure. Functional diversity falls with species diversity as N-fixers such as foliose lichens and mycorrhizal fungi, and insect and pest predators are lost (Perry, 1989; Swanson and Franklin, 1992). Without CWD to slow sedimentation, erosion and peak stream flows increase (Swanson, Franklin, and Sedell, 1990; Swanson and Dyrness, 1975). Spatial heterogeneity is reduced under the regenerating even-aged stand, and will not return to old-growth levels until gaps begin to form and shade tolerant trees become established, a process that may take 180 to 200 years (Franklin et al., 1981; FEMAT, 1993).

Traditional management disrupts the ecosystem at the landscape level as well. Homogenization is one important theme. Instead of the patchy mosaic left by natural fires, clear-cuts introduce uniformity of patch size and structure (Swanson, Franklin, and Sedell, 1990). Fire suppression further decreases variability across the landscape (Morrison and

Swanson, 1990). Most importantly, clear-cutting and short rotations place an unprecedented proportion of the forest in age classes younger than 100 years (Swanson and Franklin, 1992). A 1984 study showed that 85% of all trees in western Washington were less than 5 inches diameter at breast height (Harris, 1984).

A second important theme in traditional management is increased disturbance. The logging road network greatly increases the frequency of landslides, contributing to stream sedimentation (Swanson and Dyrness, 1975). Fragmentation of intact forest not only isolates populations by decreasing connectivity (Pickett and Thompson, 1978; Harris, 1984), but reduces the effective area of intact forest. This occurs because "edge effects" significantly alter microclimate and tree densities up to 140 meters into the forest from the clear-cut boundary (Chen, Franklin and Spies, 1992). The dispersed, or checkerboard, cutting pattern used on federal lands in the Pacific Northwest actually maximizes the amount of edge per area harvested (Franklin and Forman, 1987). The likelihood of disturbance by pathogens and, especially, wind is much higher along these edges than in intact forests (Swanson, Franklin, and Sedell, 1990). The ultimate effect is a forested landscape on the very boundary of the hypothesized range of natural variation.

"New Forestry," inspired by these recent findings, represents a broad effort to change forest management. Jerry Franklin of the University of Washington, the most active proponent of New Forestry, writes, "The term New Forestry has been used to identify the concept of using ecological principles to integrate better environmental and commodity values at the stand and landscape levels" (Franklin, 1993b: p. 138). At the landscape level, Franklin calls for increased attention to riparian zones and patch arrangement. The forest matrix, or the managed forests between reserves, should provide small scale habitat, buffer zones around reserves, and connectivity between reserves (Franklin, 1993). At the stand level, mixed tree species, including hardwoods, are preferable to monocultures; higher levels of CWD and snags should be maintained; canopy closure should be delayed to provide habitat for early seral plants and animals; and, finally, some green trees should be retained.

Green tree retention, combined with retention of natural snags and CWD, is simply an effort to maintain higher levels of structural complexity because of its correlation with biological and functional diversity. A harvest unit where green trees, snags and downed logs are retained imitates a natural disturbance much better than clear-cuts do. According to the hypothesis, these biological legacies provide habitat that allows many species to persist, and allows others to return more quickly. Where green trees are retained, spotted owls may return in 70 to 80 years, and true old-growth conditions could return two or three times faster than in a clear-cut (Franklin, 1993). Retained old-growth trees may function as refugia for foliose lichens, invertebrates, and soil fungal associations, and might inoculate the regenerating class in the future (Franklin, 1993). These stands could improve connectivity between reserves, depending on the density of retained trees and the requirements of particular organisms. Along with these ecological benefits, moderate level green tree retention remains a viable method of wood production (Franklin, 1993).

The number of green trees that need to be retained depends on the management objectives for a given stand. The range of objectives and harvest levels forms a continuum, what Franklin calls the "gradient-of-retention concept" (Franklin, 1993: p.11). At one extreme, a high level of retention can preserve original wildlife populations and maintain ecosystem functions such as erosion control. At the opposite extreme, units managed for intensive wood production will have very low levels of retention. Moderate levels of retention might accelerate re-creation of owl habitat. Different objectives determine whether trees should be retained individually or in patches. Current recommendations focus on the low to moderate end of the retention gradient. FEMAT guidelines call for 15% retention (FEMAT, 1993). Franklin describes anything from 10 to 40% as "green tree retention," and notes that 10 to 18 trees per acre may be necessary to ensure an adequate supply of snags and CWD, and 20 to 40 trees per acre would be required to create mixed-structure forests suitable for late-successional species (Franklin, 1992).

Although green tree retention could be applied in any region, the unique attributes of Pacific Northwest forests provide the strategy's rationale. The importance of structural complexity to organisms and processes and the size and longevity of the structural elements are not common to other forests. While green tree retention could be used elsewhere to create mixed-aged stands, or provide specific habitat, it is unlikely that the strategy would be so potentially critical to the regional ecosystem.

Windthrow

Windthrow affects both natural and managed forests. Research efforts have focused on understanding windthrow as an ecological process and limiting its impact on timber production. In the case of green tree retention, preventing windthrow may preserve both ecological and commodity values. However, the prevention of windthrow requires an understanding of the factors that cause it. Some of the factors influencing windthrow are common to all forests, such as wind behavior, the effect of silvicultural activities, and the concept of windfirmness. The importance of other factors varies from region to region, as illustrated by the contrast between studies in Europe and the Pacific Northwest.

Winds are profoundly influenced by topography. Obstacles compress wind into a smaller space, forcing it to increase velocity. Winds speed up as they pass over ridge-tops and upper slopes, around the shoulders of mountains, through gaps and saddles, and through narrowing valleys (Gratkowski, 1956; Moore, 1977). Landforms can alter wind direction at both large scales and scales as small as forest edges (Moore, 1977; Gloyne, 1968). Lee flow occurs when very strong winds pass over gentle ridges and the airflow remains attached, flowing down the leeward slope and further increasing in velocity (Gratkowski, 1956). More commonly, lee waves, or eddies, form when wind accelerates over a ridge, creating turbulent winds high on the leeward slope that are often more damaging than steady winds of higher velocity (Ruth and Yoder, 1953). Studies in both

Europe and the Pacific Northwest have observed more damage on leeward than windward slopes, especially in mountainous terrain (Gloyne, 1968; Ruth and Yoder, 1953). Damaging turbulence can also be generated by very small features such as knolls or even forest edges (Moore, 1977). The effect of turbulence demonstrates that, in addition to wind velocity, gustiness and changes in wind direction may be important (Gloyne, 1968). The duration of the winds influences damage, too, for prolonged swaying loosens roots in the soil (Oliver and Mayhead, 1974).

The relationship between silvicultural activities and windthrow is fairly constant from region to region. Before the turn of the century, foresters were aware that cutting openings in the forest would cause wind damage (Rothrock, 1898). Partial cuts were later observed to have the same effect. In both eastern and southwestern Oregon, up to 25% loss of volume was observed in 15-to-30 year old selection cuts (Weidman, 1920; 1920b). Thinning has the same effect, leading to 22% losses in studies in Canberra and Northern Ireland (Savill, 1983). Removing one tree can double the forces on adjacent trees (Savill, 1983). Forest edges exposed by timber harvesting suffer similar damage (Gratkowski, 1956; Savill, 1983). A survey of 1-to-6 year old streamside buffer strips in the Oregon Coast Range found that 0-to-72%, of the initial live tree basal area was windthrown, and wind damage was 20% or more at 13 of the 30 sites (Andrus and Froehlich, 1986). The size of the harvest unit does not appear to affect damage severity (Gratkowski, 1956; Moore, 1977), and whether the density of trees in partial cuts has an affect is unclear (Savill, 1983). Damage in stands where green tree retention is applied should fall within the ranges observed in these studies.

If this sort of damage continued, eventually the entire forest would be lost. In almost all cases, however, the damage is limited to the first years after harvest (Ruth and Yoder, 1953; Gratkowski, 1956). The Oregon selection cut studies found that the greatest losses occurred in the first 2 to 3 years, and all damage was concentrated in the first 6 years after harvest (Weidman, 1920; 1920b). The danger of windthrow caused by thinning decreases after 2 to 5 years (Savill, 1983). Buffer strips on Vancouver Island were also observed to stabilize after 5 years (Moore, 1977).

Studies of tree physiology confirm that individual trees become more windfirm. Because wood and roots are weaker in compression than tension, the lee side trunk and roots fail first (Mergen, 1954). Shaking and stress prompt the development of thicker trunks and increased wood and root growth on the lee side (Savill, 1983; Ruth and Yoder, 1954; Mergen, 1954). Trees growing on ridges, or in open areas exposed to wind, are extremely stable because of these adaptations (Ruth and Yoder, 1954). But trees that have grown in a dense, protected stand are extremely vulnerable when suddenly exposed to wind following harvesting. Often it is the first storms of the winter after harvest, not the most severe, that do the most damage (Moore, 1977). The field studies indicate that windfirmness increases quickly, enabling trees to resist severe storms only five years after an opening is created.

Soil type and root condition constrain windfirmness. Virtually all studies of windthrow note that wet, saturated soils offer poor anchorage and increase windthrow. When strong winds are accompanied by precipitation, the problem is exacerbated. Along with poorly drained soils, shallow soils that limit vertical rooting create the most unstable conditions (Ruth and Yoder, 1953; Mergen, 1954; Gratkowski, 1956; Savill, 1983). The one exception may be thin soils overlying fractured rock, permitting limited but secure rooting (Moore, 1977). Any root rot, pests, or physical damage from fire or logging that affects roots will increase the risk of wind damage (Ruth and Yoder, 1954; Savill, 1983). When root strength is low, or the soil is saturated, the tree will uproot; if the roots are anchored firmly, the trunk may break.

However, the relative importance of soil type and other site factors in determining windthrow risk varies significantly between different regions and forest types. In Europe and New England, conditions are predictable enough to allow researchers to make wind risk classifications. European models are based primarily on elevation, aspect, and soil type (Savill, 1983). Systems in Britain and Northern Ireland include an exposure index rating (Lowe and Keane, 1991). Research on hurricane damage in the northeastern United States found elevation and aspect to be important, along with tree species and size (Foster, 1988). In these regions, a few site factors emerged as important determinants of windthrow.

Studies in the Pacific Northwest have not been able to produce reliable windthrow risk classifications. A study of buffer strips on Vancouver Island found that windthrow "is caused by the simultaneous interaction of a number of natural factors including location, local topography, climate, aspect and slope, soil depth and texture, tree species and rooting characteristics, and stream characteristics" (Moore, 1977: p. 7). The importance of complex interactions means that distinct site factors have little predictive power. Andrus and Froehlich (1986), working on stream buffers in coastal Oregon, identified four significant site factors that increased the risk of windthrow: the proportion of the stand on boggy terraces, the proportion of conifers in the stand, the orientation of the stream with respect to prevailing southwesterly winds, and the site's relative exposure. Their equation explained 57% of between site variation. Since two of these variables are specific to buffer strips, only the proportion of boggy terrain and relative exposure apply to green tree retention cutting units. Accurate windthrow risk assessment for potential harvest sites remains difficult.

Despite the importance of interactions and variation in the Pacific Northwest, some regional generalizations about windthrow can be made. Poorly drained soils increase damage, especially since the highest winds occur during winter storms when heavy rain is common (Ruth and Yoder, 1954; Moore, 1977). Damage is often worse closer to the coast (Franklin et al., 1987; Moore, 1977). Since storm winds in the Pacific Northwest consistently blow from the south through the west, some studies have observed more catastrophic damage on leeward, north and east facing, slopes (Ruth and Yoder, 1954). However, severe easterly foehn winds do occur, especially in large east-west trending valleys such as the Columbia River Gorge. Cedars are the most windfirm species, followed by Douglas-fir, Western hemlock, and Pacific Silver-fir (Gratkowski, 1956). This may have more to do with other site factors that correlate with tree species than with the properties of the trees (Moore, 1977).

Study objectives

Identifying the proper levels of green tree retention for given objectives and locations depends on an understanding of windthrow. The first objective of this study was to describe patterns of windthrow and other forms of mortality, measured 1 to 10 years after harvest in cutting units where green tree retention was applied. The second objective was to identify site factors determining windthrow. If strong predictors of wind-risk emerge, prescriptions for green tree retention in the future could be adjusted on a stand-by-stand basis. If accurate prediction of windthrow proves difficult or impossible, then the implications of green tree mortality must be evaluated at the landscape level.

Study area description

All cutting units surveyed are located on the Blue River Ranger District of the Willamette National Forest, Oregon, between 1600 and 4400 feet elevation. The Western Cascades were formed by volcanic activity and modified by some glaciation. The chief parent materials are pyroclastic rocks, andesite and basalt. The pyroclastic rocks produce soils with a silt to clay texture, prone to shallow rooting and instability (Moore, 1977). Basalt and andesite produce coarse textured, more stable soils (Steinblums, et al. 1984). Mean annual precipitation ranges from 50 to over 130 inches, increasing with elevation. Winters are moderate, with rain at lower elevations and snow common at higher elevations. Summers are dry and hot, and daily highs may be above 90 degrees.

At lower elevations, Western hemlock (*Tsuga heterophylla*) plant associations predominate. Douglas-fir (*Pseudotsuga menziesii*) generally forms the canopy, with Western hemlock and Western redcedar (*Thuja plicata*) in the understory. The higher elevations of the study area enter the Pacific silver fir (*Abies amabilis*) plant association.

Damaging winds typically occur during winter storms coming off the Pacific, and blow from the southwest. In the storms of 1990, the most severe since the Columbus Day storm of 1962, peak gusts were estimated to exceed 100 m.p.h. (George Taylor, personal communication, 1993). Since extreme events may not have a periodic nature, estimating a return interval involves great uncertainty. The storms of 1990 may have been anything from 30 to 100 year events (Sherwood, 1993).

Methods

Site selection and data collection

A list of timber sales with silvicultural prescriptions for green tree retention was compiled from files at the Blue River Ranger District and refined with help from experienced District personnel. The prescriptions for some older sales call for only 2 "wildlife trees" per acre. This was the minimum level of green tree and snag retention included in the study. More typical guidelines require 3 to 6 green trees per acre, in addition to existing snags that can be left safely. At the high density extreme are two shelterwoods holding 15 to 20 trees per acre.

One to three units per sale were chosen arbitrarily in the field, though an effort was made to stratify sites across aspect and elevation. Forest Service records supplied each cutting unit's elevation, size in acres, date of harvest, and history of silvicultural activities such as salvage, thinning, or, after harvest, topping. In all, 44 cutting units were inventoried in July and August, 1993. Sites that covered more than one aspect were subdivided for the windthrow analysis only, providing 48 total samples (Appendix 1).

The following site factors were noted in the field: aspect, original forest type (mature, mixed, and old growth), topographic position (ridge-top, upper, middle, and lower slope, bench), slope, and edge type (forest, clear cut, stream, road). Tree species and individual tree size were not recorded. The inventory included the number of green trees standing, broken and topped, and trees that died after harvest, as opposed to pre-existing snags, which were also recorded. Forested edges of the cutting units were surveyed for blowdown, and any damage was classed as low, moderate or severe, depending on the length of edge affected and the depth of penetration into the forest.

The distinction between snags and green trees that died after harvest can easily be made based on the extent of decay, and on the different ways dead and live trees are scarred by slash fires. Windthrown trees were located and counted, and each tree's direction of fall and type of blowdown (uproot or break) recorded. The presence of fine root hairs and conical scars from slash fires distinguish trees blown down after harvest from existing down logs characterized by advanced decay and a uniform burn pattern. To separate windthrown green trees from snags that fell after harvest, subjective assessments of decay were made, but the presence of charred wood was more definitive: If the tree had been alive at the time of the broadcast burn, typically only the bark would have scarred.

Statistical analysis

Differences in time of harvest confound the windthrow results in three ways. First, total windthrow should increase with time since harvest: A site that has been exposed for many winters should have more damage than a recently logged unit. Second, large storm events confound the relationship between time since harvest and windthrow by dramatically increasing the probability of damage in certain years. Finally, the poorly understood process of retained trees becoming windfirm may reduce the probability of damage with time.

The simplest model of windthrow, assuming no variation in severity from one winter to the next, addresses the first issue of total time since harvest:

$$\text{Trees standing now} = \text{Trees retained at harvest} * S^t$$

where S is the survivorship rate and t is the number of winters since harvest. The terminology can be confusing: *Actual survivorship* is simply the number of standing trees divided by the total number of green trees at time of harvest, ignoring the number of winters the site has been exposed. Its inverse is total, or cumulative, windthrow. *Average annual survivorship*, or its inverse, annual windthrow, is the site-specfic yearly rate referred to in the equation above. *Overall average annual survivorship* is the mean annual rate of all sites sampled.

To measure the effect of both the timing of large storms and increasing windfirmness, a model that takes into account year-to-year variation in survivorship is needed. While it is impossible to estimate the survivorship during each winter for each individual site, we can estimate an average survivorship of all sites exposed to a given winter. *Forest-wide survivorship* refers to the mean survivorship of all sites exposed to a given storm season for that one year. For example, a site that was logged in the summer of 1990 was exposed to three storm seasons before the inventory, so

$$\text{Trees standing now} = \text{Trees retained at harvest} * S1 * S2 * S3$$

where S1 is the forest-wide survivorship of winter 1993, S2 the survivorship in 1992, and so on. In more general form, taking the logarithms of all survivorships,

$$\log S = \sum_{i=0}^{t} \log Si$$

with log S the actual survivorship for the site, and t the number of years since harvest. The key is to find the Si that minimize the difference between the actual and predicted survivorship for all 48 sites. This can be solved as a linear least squares problem (Bossert, personal communication, 1993):

$$\text{Residual sum of squares} = \sum_{\text{all sites}} (\log S - \sum_{i=1}^{t} \log Si)^2$$

Some accommodations had to be made for the lack of any sites cut in years 6, 8 and 9 (see Table 1). Since the survivorships of each of these years could not be separated, S6 actually represents the combined survivorship of 1987 and 1988, and S7 1984 through 1986.

Using analysis of variance, the effect on windthrow of site factors such as aspect can be compared with the effect of timing, or date of harvest. The residual variance in the simple model can be explained perhaps by between group variances when the sites are grouped according to aspect, elevation, or other site factors. Similarly, the model that incorporates year-to-year changes in survivorship may explain some of the variation between sites harvested different years, reducing the total residual variance. The reductions in variance can be compared, showing the relative importance of the effects.

Analysis of variance assumes a normal distribution. Average annual rates of survivorship and mortality are not normally distributed, for the data points are pressed up against one or zero, respectively. If subtle distinctions are the objective, than an ANOVA is a completely inappropriate test. But in this case, where relative comparisons and qualitative impressions are the goal, the results still can be valuable.

Results

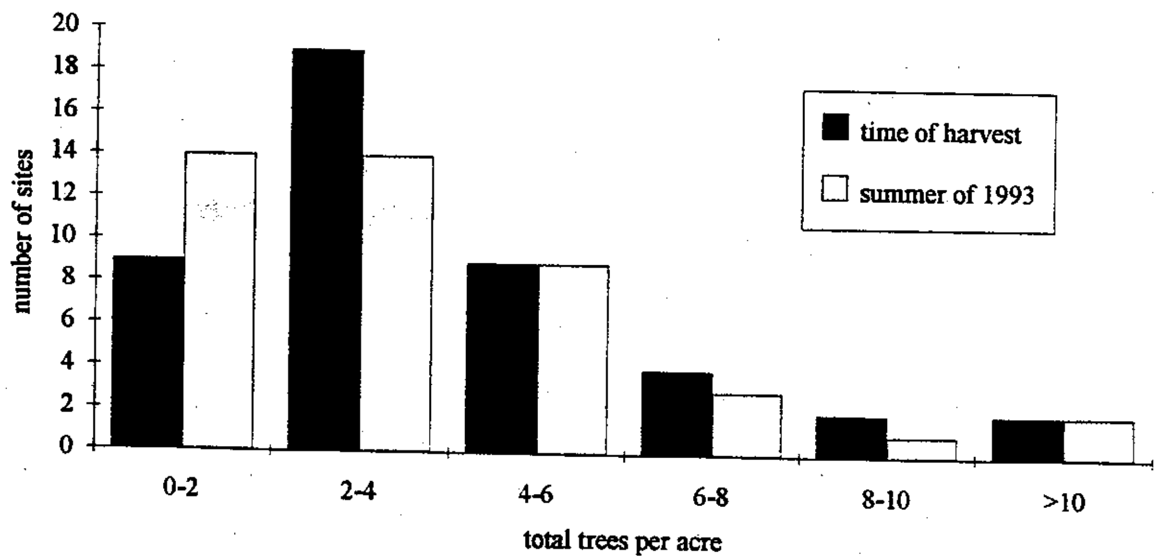
Site Characteristics

Though unit size ranges from 4 to 58 acres, most units are between 10 and 25 acres. Total tree densities (live trees and all snags) retained at harvest ranged from 1 to 4 trees per acre for more than half the sites. Five sites shift from the 2 to 4 trees/acre class to the 0 to 2 trees/acre class when total tree densities retained at harvest are compared to densities at present (Figure 1). The prescriptions apparently ignore the pre-harvest relationship between tree size and density, because former old-growth sites had the highest mean density retained at harvest, followed by mixed age stands and then mature stands, averaging 5.0, 4.4 and 3.8 total trees per acre respectively. Mean density at present,

Table1. Sample size and actual year of forest-wide survivorships in statistical model

first winter	forest-wide survivorship year	sites harvested during each year	total sites exposed
1993	S1	1	48
1992	S2	15	47
1991	S3	12	32
1990	S4	10	20
1989	S5	6	10
1988	S6	0	4
1987	S6	2	4
1986	S7	0	2
1985	S7	0	2
1984	S7	2	2

Figure 1. Frequencies of total tree densities retained at harvest and standing in summer, 1993



reflecting the slightly different amounts of windthrow in each group, follows a similar pattern: Old-growth areas average 4.2 trees per acre, and mixed and mature stands average 3.4 trees per acre.

Total tree density includes live trees, pre-existing snags, trees that died after harvest, and trees topped to create nesting habitat. The number of snags per acre retained at harvest has fallen over time (Figure 2). A simple regression found a slope of -0.127, significant at the 1% level (R^2=0.22). Total dead per acre, which adds trees that died after harvest and topped trees to natural snags, shows the same trend, only weaker. In contrast, total trees per acre retained at harvest has remained relatively constant, if not increasing, over time. Snags blown down by wind can be identified at one third of the sites, but they are rare enough to have little impact on total tree density. Live, or green, trees were "topped" at 11 sites to provide nesting habitat. Originally, this was done by climbing the tree and cutting off the crown with a saw. Most of the sites in this survey were topped more recently using explosives, leaving a broad, jagged break. A mean of 26.5% of the green trees were topped at each of these units.

Mortality

This survey recognizes only two categories of mortality: Wind, responsible for uprooted and snapped trees, and other agents, which killed the trees but left them standing. Most of the trees that fit this second category appear to have been killed by slash fires (Figure 3). Mechanical damage from logging may also have contributed to the mortality of green trees. Mortality due to both wind and other agents was extremely variable, but quite high overall. Total, or cumulative, mortality due to wind ranged from 0 to 58% at the most severely damaged site. The average actual windthrow was 15.6%, with 19% of windthrown trees broken and the rest uprooted. Other mortality averaged 15.9%, with a standard deviation of 16.8. Combined, total mortality ranged from 0 to over 80% (Figure 4).

Figure 2. The relationship between number of natural snags retained per acre and time of harvest

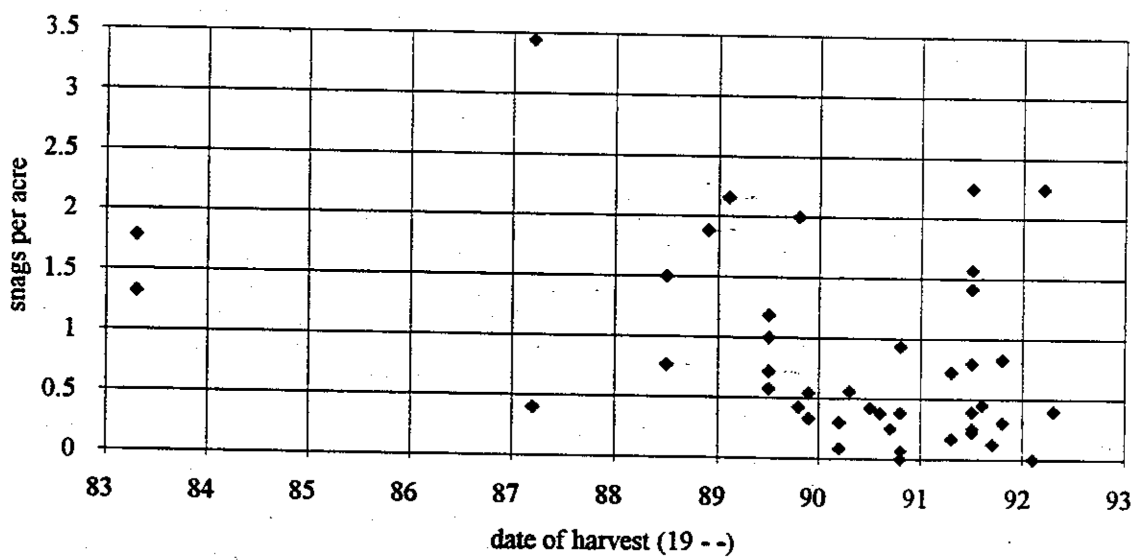
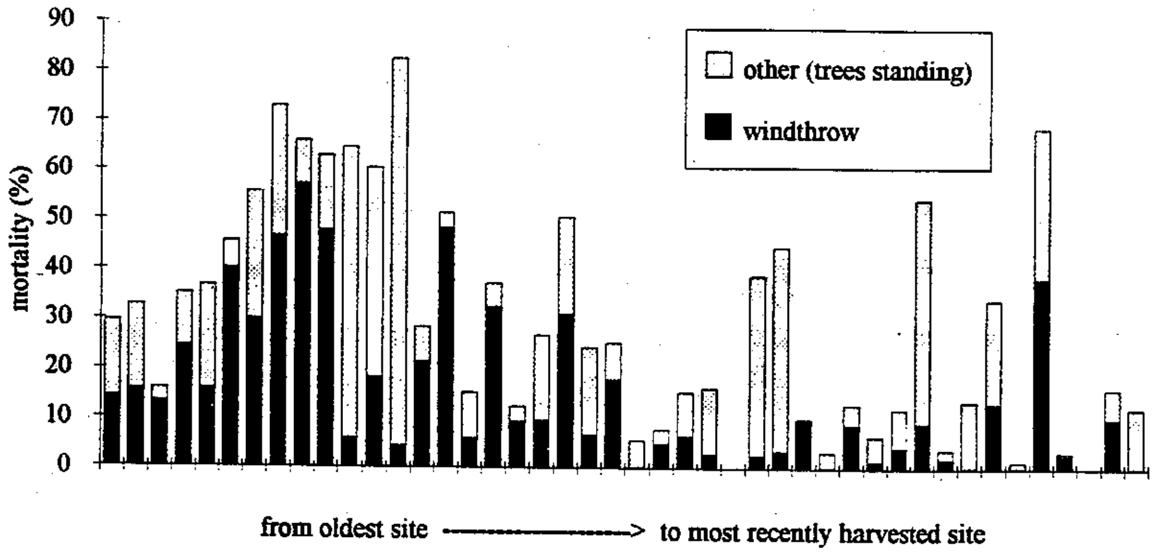


Figure 3. Total windthrow in this unit (Lytle 7) was 46.7% and other mortality, mostly caused by a very hot slash fire, was 26.4%.

Figure 4. Windthrow and other mortality of retained green trees in all 44 harvest units



Wind caused considerable damage on the edges of the cutting units (Table 2). Although the severe damage was caused by winds blowing across the unit into the leeward edge, at some sites trees were blown down *into* the unit, evidence of turbulence as the wind dropped down from the intact canopy on the windward side. At 8 sites, blowdown was severe, extending at least three rows of trees into the forest along a significant length of one edge. Moderate damage, slightly more common, was not as continuous along the edge nor did it penetrate as deeply into the forest. Many sites showed low damage, meaning that individual trees scattered along the edge were windthrown. Only ten sites suffered no blowdown on any forested edge.

Edge types varied considerably in vulnerability to windthrow (Table 2). Edges placed on stream buffers were particularly unstable. Both naked buffers, extending into the center of a unit along both sides of a stream, and true edges alongside streams had two cases each of severe windthrow. But while 5 of the 6 naked buffers surveyed suffered at least moderate damage, only 3 of 7 buffers on the edge of the unit had comparable blowdown. Forest edges on slopes--the vast majority of edges surveyed--had relatively less damage than stream-side or wet areas, while ridge-top edges suffered no severe damage.

Windthrow analysis

The total, or cumulative, windthrow presented above ignores the number of years since harvest. The average annual windthrow at each site, which ranges from 0 to 22.6%, with an average of 4.7%, is a better measurement for comparison. Variation is high, with a standard deviation of 5.7. Average annual windthrow does appear to be related to time of harvest: The mortality rates in units harvested the year or two preceding the winter of 1990 are higher than sites created after 1990, and also higher than the oldest units (Figure 5).

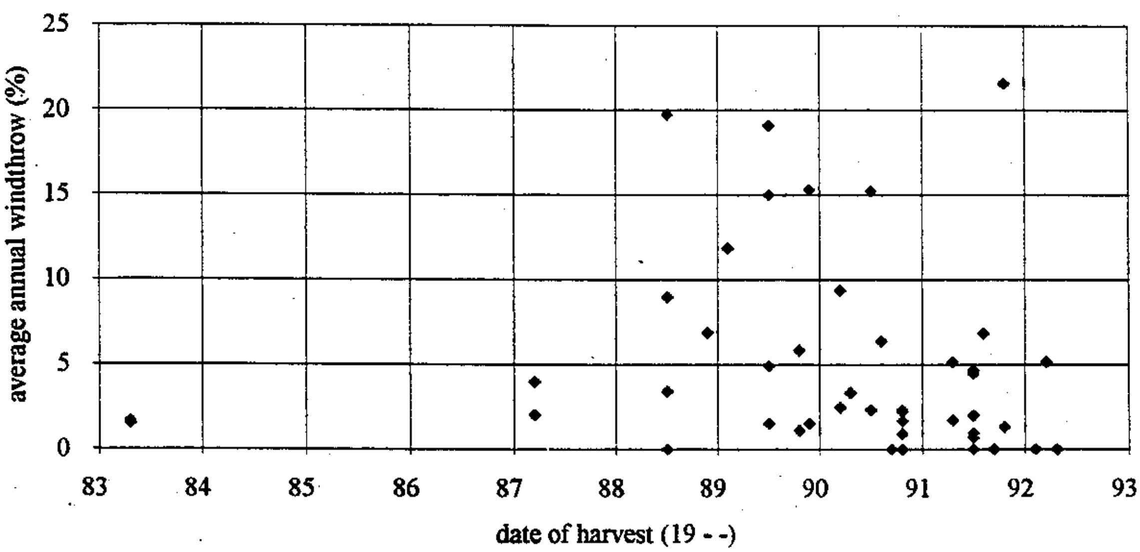
Such high site-to-site variation makes predictive modeling extremely difficult. When the sites are grouped by either aspect, elevation, topographic position, forest type,

Table 2. Windthrow severity along edges grouped by edge type

	Windthrow Severity			
edge type	none	low	moderate	severe
naked buffers*	1	0	3	2
stream buffers	3	1	1	2
ridgetops	9	5	2	0
forested slopes	67	27	4	4
clear-cuts	44	na	na	na
Totals	214	33	10	8

* stream buffers that extend into the middle of the unit, exposd on both sides

Figure 5. The relationship between each site's average annual windthrow and time of harvest



soil type, or stand history, differences in the mean average annual windthrow of each group do appear, but large standard deviations make these differences statistically insignificant (Table 3). For example,with respect to aspect, the mean average annual windthrow of sites facing north and west was almost twice as high as sites facing south. Aspect would appear to be the best site factor for predicting vulnerability to damage from wind. But moving from the simplest model of windthrow to one that incorporates aspect groups reduced the residual variance insignificantly, less than 4%. Since the variation between aspect groups explains so little, much variation must exist within each group.

To incorporate time of harvest into a model requires the linear least squares equation explained previously. By comparing the windthrow at sites harvested in different years, it is possible to estimate forest-wide survivorships for each winter (Table 4). Survivorships greater than 1 in the oldest years are impossible biologically, but not mathematically. The only way for the computer to account for the relative lack of damage in the oldest sites, compared to younger sites which also weathered the storms of 1990, was to push up the survivorships in those first years. A second run, which assumes that trees become windfirm after the first year (survivorship in the first year is squared), homogenized the results, lowering the survivorships of the highest years and raising the lowest rates.

An analysis of variance for time of harvest was performed. The model that considers year-to-year variation in survivorship, compared to the simplest model, lowered residual variance 24%. The F-statistic, at 6 and 41 degrees of freedom, is 2.16, significant at the 10% level. Both sets of forest-wide survivorships, with or without windfirmness, produced virtually identical results.

Aspect, which appeared to be the most promising site factor for predicting windthrow, explains very little variation. Time of harvest proved to be a much better indicator of windthrow damage and the most important site factor included in the survey. Since time of harvest has a marginally significant effect, a more appropriate test would be required to confidently determine its statistical significance.

Table 3. Standard deviations and differences in average annual windthrow (%) between site factor groups

Aspect	N	E	S	W	
mean	6.0	3.9	3.5	5.6	
standard dev.	7.5	4.1	6.1	5.0	
Topo. position	lower slope	mid slope	upper slope	ridge	bench
mean	3.5	3.4	7.1	4.4	5.2
standard dev.	3.7	3.5	7.5	5.7	8.0
Soil type *	green breccias	deep till	shallow red breccias	deep red breccias	fractured rock
mean	1.7	7.8	6.3	4.6	2.9
standard dev.	1.7	6.8	6.9	4.7	4.6
Elevation (ft.)	<2000	2000-3500	3500<		
mean	4.2	4.8	5.1		
standard dev.	4.7	5.6	7.0		
Forest type	mature	mixed	old-growth		
mean	3.9	5.9	5.0		
standard dev.	4.9	6.5	6.3		
Stand history	disturbed	undisturbed			
mean	4.1	5.1			
standard dev.	4.5	6.3			

*Legard and Meyer, 1973.

Table 4. Forest-wide survivorships estimated by the statistical model

actual year	model year	estimated survivorships	survivorships assuming windfirmness
1993	S1	1.00	1.00
1992	S2	0.926	0.962
1991	S3	0.981	0.972
1990	S4	0.794	0.879
1989	S5	0.879	0.879
1987-88	S6	1.274	1.058
1984-86	S7	1.053	1.056

The importance of interaction between site factors is illustrated by the relationship between aspect and wind direction, indicated by the direction of fallen trees. Winds coming from the south through the west accounted for 59% of all windthrown trees. But within each aspect group south and west winds were not always the most damaging. Downslope winds were responsible for a disproportionate amount of damage. Southerly winds were most damaging on north facing sites, while north winds were most damaging on south facing sites. On east facing sites, west winds did over five times more damage than east winds, and vice versa on west facing sites. A table of expected frequencies, based on the null hypothesis that winds from a given direction will account for the same proportion of damage on all aspects, was compared to the observed frequencies (Tables 5 and 6). The chi-squared goodness-of-fit test was highly significant for south, southwest, west, east and southeast winds, meaning that each of these winds were far more damaging on some aspects than others.

Table 5. Damage classed by wind direction and aspect, observed frequencies

Number of trees felled by winds from the

		S	SW	W	NW	N	NE	E	SE	Total
Aspect	N	32	30	30	3	3	5	6	4	113
	E	11	19	22	4	9	3	5	3	76
	S	17	31	38	10	13	9	7	5	130
	W	13	4	4	7	10	8	35	25	106
	Total	73	84	94	24	35	25	53	37	425

Table 6. Damage classed by wind direction and aspect, deviations from expected frequencies

Number of trees felled by winds from the

		S	SW	W	NW	N	NE	E	SE
Aspect	N	12.6	7.7	5.0	-3.4	-6.3	-1.6	-8.1	-5.8
	E	-2.1	4.0	5.2	-0.3	2.7	-1.5	-4.5	-3.6
	S	-5.3	5.3	9.2	2.7	2.3	1.4	-9.2	-6.3
	W	-5.2	-17.0	-19.4	1.0	1.3	1.8	21.8	15.8
	p	0.1	0.01	0.01				0.01	0.01

Discussion

Composition of retained trees

Live trees, windthrows, natural snags, topped trees, and trees killed by fire or logging damage all perform different ecological functions. The quantity of each component varies from site to site and through time. For example, the number of natural snags retained per acre fell over time. This trend may be an artifact of sampling. Three of the four units harvested before 1988 had much higher than average snags per acre, possibly because in old sites there is a greater tendency to mistake trees that died after harvest for natural snags. Also, total tree densities were above average in these three particular sites. Finally, the number of snags left per acre is related to forest type. In old-growth units, an average of 0.62 snags were retained per acre, compared to 0.48 in mixed stands and 0.27 in sites with mature forests. The sampled old-growth units were, on average, harvested earlier than mixed and mature stands, a trend compatible with the increasing scarcity of old-growth forests. Fewer natural snags in the original stand means fewer snags can be retained.

Since fewer natural snags are being retained, it is important to know whether topped trees and fire-killed trees are viable substitutes, especially in younger stands where snags are less abundant. Trees killed by fire form "hard" snags, as opposed to the "soft" snags created by disease and used by cavity nesters. Topped trees offer a unique type of nesting habitat as well. In sites where trees were topped, the combined mean density of topped trees, green trees that died after harvest, and natural snags, is 1.1 per acre. This figure should be compared with the known potential population densities of targeted wildlife species.

Whether or not prescriptions have changed since the mid 1980's, the tree composition within each unit certainly changes over time, as the high rates of windthrow and other mortality attest. From an ecological point of view, how these changes are distributed through time is critical. If windthrow can be expected for decades, it would be an important agent of the stand's structural development. Large trees would come crashing down on the regenerating class, creating gaps in the future canopy. But if retained trees do become windfirm and damage from wind only occurs soon after harvest, before significant regeneration, windthrows will not have the structuring effect of later tree-falls. The large, downed logs of windthrown trees will perform important ecological functions, but this addition will come as a single pulse. The literature suggests that this latter scenario is more likely, a hypothesis supported by the relative lack of wind damage in the survey's oldest units compared to sites harvested just before the severe winter of 1990.

Mortality from fire and logging also should be viewed as a single disturbance, rather than an ongoing process. Unlike windthrow, management can effectively address these sources of mortality with simple changes in logging prescriptions and slash disposal practices. Perhaps a longer study of green retention trees could identify other causes of mortality that are known to operate in natural forests.

Predicting windthrow

The quantity of windthrow found in this survey is consistent with previous studies. Windthrow after thinning or partial cuts has ranged from 20 to 25% (Weidman, 1920; Savill, 1983), and from 0 to 72% in streamside buffer strips in coastal Oregon (Andrus and Froehlich, 1986). The range of 0 to 58% actual windthrow in this survey, and the average of 15.6%, fall within precedent.

But while Andrus and Froehlich identified four significant stand or site characteristics, this survey failed to isolate any predictive factors. The best determinant of wind damage was time of harvest, significant at the 0.10 level. Since retained stands appear to become windfirm quickly, the timing of large storms is critical: If a unit can survive its first few years without great damage, it may never suffer considerable damage. However, if a large or even moderate storm hits the site soon after harvest, damage will be high. Understanding the importance of timing, however, is little help in predicting the likelihood of windthrow at a given site.

The failure to identify predictive factors has three reasonable explanations: 1) The survey did not include the critical variables; 2) the sample size was too small and the variance too high to produce statistically significant results; 3) The region's dissected topography and complex interactions between so many site factors prohibit accurate wind-risk prediction. The first two answers tend to treat the high variance as an artifact of sampling, while the last answer offers an explanation for the great variance.

The first explanation, that the critical variables were not measured, can probably be disregarded because of the high variances. Admittedly, tree species and size, not included in the survey, were found to have an effect in previous studies in the region. A better soil classification system could have been used. But no matter which variables were measured, the extreme site-to-site variability would remain a problem. The importance of within group windthrow variance relative to between group variance would prevent significant results.

The second explanation also addresses the problem of sampling. A larger sample size would have increased the effect of site factors relative to variation. With a large enough sample size, the difference in windthrow on north and south aspects would be significant, assuming the means hold constant--a very uncertain assumption. In fact, the mean windthrow for each aspect group did not hold constant over time. Units exposed to the storms of 1990 averaged 6.8% average annual windthrow on north aspects, 5.3% on east, 7.4% on south and 7.9% annually on west aspects. Units not harvested until after the winter of 1990 averaged 5.3% on north slopes, 2.6% on east, 0.8% on south, and 4.1% on east slopes. While damage was more evenly distributed across aspect in the older cohort, large differences in damage by aspect emerge among recently created sites. More importantly, the pattern changes dramatically: west and south aspects suffered the highest windthrow in the older sample, but north aspects experienced the highest windthrow in the recent sample. Just as overall damage is highly variable from year-to-year, it seems that damage from aspect-to-aspect will also vary significantly with time.

It seems unlikely that such variation is an artifact of sampling. In fact, the great variation, and not the differences between group means, is the trend that requires explanation. The last explanation treats variation as a product of the unique environment of the Pacific Northwest. As discussed earlier, previous wind-risk classification efforts in the region were also hindered by complex interactions between many factors, and generated subjective guidelines, not quantitative regression equations. These classifications rely on a good deal of common sense and local experience. Moore (1977), working on Vancouver Island, proposes four steps towards a *relative* assessment of blowdown likelihood: 1) assess general geographic location and orientation of the site in relation to expected storm winds; 2) determine if the local topography will funnel winds or create turbulence over the site; 3) examine "soil depth and texture, tree species and rooting characteristics" and search for evidence of previous windthrow (p. 27); 4) consider the likely impacts of damage. Andrus and Froehlich write, "We still face great uncertainty in deciding whether a proposed buffer strip is likely to suffer wind damage or not" (1986: p. 10).

Forecasting windthrow in the Cascades may be even more difficult than in coastal forests or landscapes with low relief where wind direction is more predictable. The steep, dissected terrain of the Cascades funnels winds through valleys and around obstacles at large and small scales. Pockets of turbulence can form behind any of the countless ridges and spurs. In geographic locations where winds come off water, or cross relatively flat terrain, the interaction between landscape and wind behavior may not be as important.

A case-by-case approach illustrates the difficulty of assessing damage potential. The site with the highest average annual mortality, 21.6% (Lytle 1), was harvested in 1991, after the bad winter of 1990. The damage is concentrated on a flat, wet area. Apparently, the saturated soil was the operating factor, not the unit's exposure to winds funnelling down Quartz Creek and Lytle Creek, since downslope winds from the ridge well to the east caused much of the damage. The unit with the second highest mortality rate, 19.1% annually (Ennis 1), was harvested in the summer of 1989 and is near the Mackenzie River. Many of the windthrown trees were blown down by winds crossing a large clear-cut to the west, while trees on the east edge of the unit, facing north, were damaged by downslope winds. The clear-cut to the west and the constriction of the river valley by the bench can be blamed for some of the mortality, but damage by winds from the south and the high level of damage overall are difficult to understand. In contrast, mortality in the unit ranking third is easily explained (Titan Too 2). The unit, facing east, sits just below the summit of a north-south trending ridge. Two high-points on the ridge funnel west winds directly downslope into the unit, which has a small creek running through its center. The unit was logged in the winter of 1990, in the midst of severe storms. Yet this unit suffered less damage than the previous two, losing 15.3% of the retained trees per year. If explaining past windthrow in existing units is so difficult, forecasting damage for prospective sites will be all but impossible.

The vulnerability of freshly exposed trees may increase variation as well. Since windfirmness is so low for retained trees soon after harvest, damage can occur virtually anywhere. Very small pockets of turbulence that might not affect an intact forest may cause considerable windthrow of retained trees and harvest unit edges. In natural forests, "endemic" windthrow is a continuous, low intensity disturbance, while "catastrophic" windthrow describes high damage from a single event (Savill, 1983). However, in recently harvested green tree retention units, perhaps all winds, endemic and catastrophic, cause severe damage. According to this hypothesis, the vulnerability of retained trees determines windthrow more than site factors such as aspect or soil type.

Despite uncertainty, the effort to predict windthrow from site-to-site should not be abandoned. The common sense approach that Moore describes would have anticipated severe windthrow in the boggy site described above (Lytle 1), and the site where winds were funnelled over a ridge (Titan Too 2). In these locations, many more trees should have been retained. One step that should be added to Moore's risk assessment is consideration of turbulent, downslope winds. On a north facing slope, exposure to the north is less important than the topography to the south: How far above the unit is the nearest ridge, or turbulence inducing feature? Average annual windthrow on upper slopes was almost 2% higher than on any other topographic position, an important if not statistically significant result. Although a formula for predicting damage may be impossible, a simple awareness of relative windthrow likelihood could prevent some of the worst damage and could improve the chances for green tree retention to meet its stated objectives.

Simulating future damage

An alternative to the site-by-site approach is to use the survey's results as an expected range of damage over a population of units. Managers can set prescriptions for green tree retention based on this expected range of mortality. But the survey gives a snapshot of sites only 1 to 10 years old; management decisions must be based on a much longer time span. How many trees will be standing after twenty, fifty or one hundred years? If the mortality rates observed in the survey extend over longer time periods, the damage is tremendous, and probably unacceptable. Taking the overall average annual windthrow rate of 4.7%, after 20 years, 38% of retained trees will remain, and after 50 years, less than 9% will still be standing.

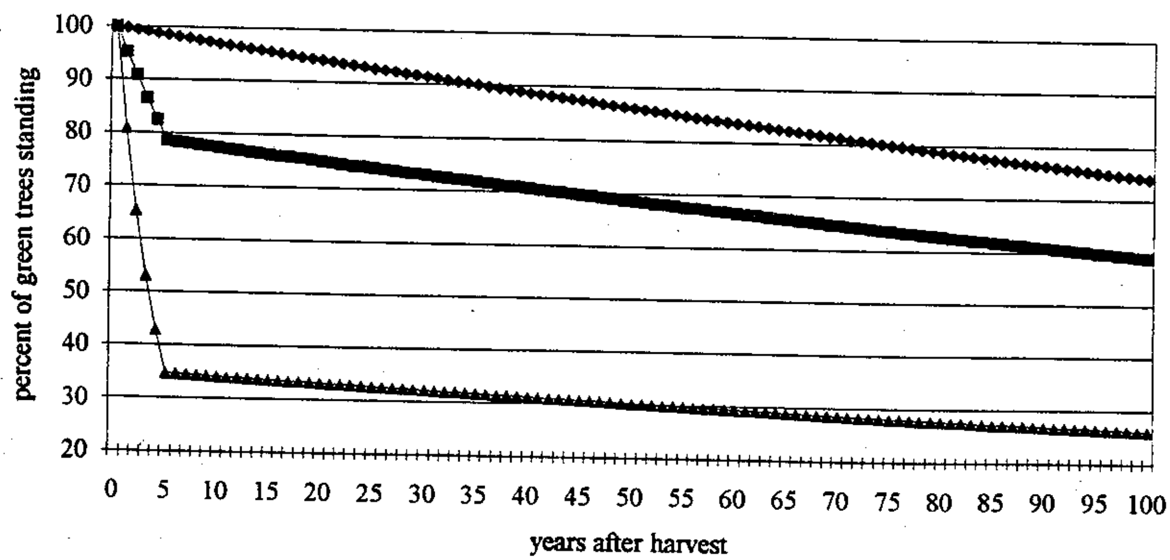
It is unlikely that the windthrow rates will remain as high because the retained stands should become windfirm quickly, an assumption supported by the literature and by the year-to-year variation found in the study. One hypothesis is that the long term windthrow rate will approach the level observed in the surrounding undisturbed forest. At the H. J. Andrews Experimental Forest, in the center of the study area, annual mortality in Douglas fir-Western hemlock forests is 0.7%, 33% of which is due to wind (Franklin et al., 1987). After the first 5 years, it may be assumed that the annual windthrow in all sites will approach 0.3%. The best case scenario for the first 5 years would be no damage beyond this baseline mortality. The worst case scenario could be the 19.1% average annual windthrow observed at Ennis 1, treating the much higher rate at recently harvested Lytle 1 as an outlier. In between these two extremes is the overall average annual mortality of 4.7%. When these three possible scenarios are projected over 100 years, from 25% to 75% of the original trees remain standing, with an average of about 60% survival (Figure 6). In terms of tree density, assuming 10 trees/acre were retained, the range runs from 2.5 to 7.5 after 100 years, and the overall average rate gives just under 6 trees/acre. Since most of the damage is done the first five years, the time horizon chosen does not make a dramatic difference. Depending on which scenario is chosen, only about 5-12% of the original stand is lost between year 50 and 100.

Implications for management

The implications of such high mortality depend on the defined objectives of green tree retention. If retained trees are to serve primarily as habitat for wildlife, specifically cavity nesters, then many trees must remain standing. From this point of view, the worst case scenario is far more important than either the likely range of damage or the average survivorship. Using an arbitrary example, if the wildlife manager wants to ensure that 10 trees per acre remain in every site after 100 years, then many more than 10 trees per acre must be retained. In fact, the potential of 75% windthrow means that the number of trees retained must be four times the minimum acceptable level, so retention of 40 trees per acre would be necessary. Less stringent criteria might require that the minimum density be met in a only certain percentage of units, and prescriptions could be set accordingly.

From a broad ecological point of view, whether the retained trees survive or are blown down is less important. A standing tree and a large, down log both provide a piece of the structural legacy that separates old-growth from younger forests. Windthrow in riparian zones may even be desirable, given the importance of coarse woody debris to stream habitat (Franklin, 1992). In general, the more of these elements left in a managed stand, the better it imitates a natural disturbance. Should green tree retention be applied frequently enough that these units comprise a significant portion of the landscape, then a

Figure 6. Expected long-term survivorship for sites with severe, average, and low windthrow



range of conditions would be preferable to artificial homogeneity. Some units will have few standing trees and many down logs, others will have a high density of standing trees, living and dead. In contrast to a strict wildlife objective, the variation in the amount of retained structural elements and their distribution across the landscape is more important than the minimum tree density in each unit.

Emphasizing the contrast between the wildlife manager's point of view and the forest ecologist's may be misleading. After all, levels of retention and corresponding management objectives form a continuum. Most of the sites included in this survey are "clearcuts with structural retention," falling on the low end of Franklin's retention gradient (Franklin, 1993: p. 12). The possible management objectives for such units include maintenance of minimum levels of coarse woody debris, a low probability of recreating owl habitat and, primarily, intensive wood production. Windthrow will not adversely effect either CWD or commodity production, but will undermine the potential of these sites to offer habitat for owls and similar species. The results of the survey suggest that the retention levels prescribed for a given management objective must be adjusted to accommodate the windthrow of 25 to 75% of retained trees in the long-term.

Databases

The most difficult part of the survey was collecting site information at the Forest Service district office. Some information, such as silvicultural prescriptions and soil type, are not included in the computer databases and were found in the original files by hand. Information that has been computerized often is listed in different databases, making access difficult. Continued monitoring of green tree retention units *with* an organized and complete database is essential.

Conclusion

Future levels of green tree retention should reflect the likelihood of high mortality at many sites in the first years after harvest. Efforts to adjust prescriptions should focus on defining a preferred range of conditions over the landscape, rather than identifying the factors which predict windthrow risk at individual sites, for two reasons: First, great variation overall and the complex interactions of many factors hinder prediction, and, second, year to year variation in mortality, or the timing of large storms, appears to be a more important determinant of windthrow than any single site factor. As long as the weather remains unpredictable, so will wind mortality in each unit. Until wind risk classifications improve, the number of green trees retained must be much higher than the long-term minimum acceptable level. Caution is the prudent response to uncertainty. Continued monitoring of these sites over longer time periods is necessary to understand the process and rate of retained trees becoming windfirm.

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Appendix 1. Site factors and tree inventory data

Sale name and unit #	date of harvest	forest type*	elevatn.	aspect	topo. position+	soil resource index	slope	unit size (acres)	silvic. Rx	total trees per acre	avg. annual windthrw (%)	green trees	topped trees
Mendel 1	83.3	OG	2200	N	3	23	17-35	22		4.3	1.51	55	0
Mendel 2	83.3	OG	2200	S	3	212	30	32		4.8	1.67	78	0
Wildwood 2	87.2	MAT	3200	N	5	233	5-25	10	12+	15.1	2	112	15
Wildwood 4	87.2	MIX	3300	E	4	233	15-20	9	12+	20.4	3.95	111	0
East Isolation 15	88.5	MIX	2700	N	5	210	30	4		5.3	3.38	12	0
Elindel 15.1	88.5	MIX	2800	W	5	16	15-30	5	2	3.6	8.97	10	0
Elindel 15.2	88.5	MIX	2800	S	5	16	0-35	5	2	3.0	19.74	4	0
Elindel 15.3	88.5	MIX	2800	N	5	16	40	3	2	3.3	0	5	0
Lytle 3	88.9	OG	3000	S	3	213	30-40	17	4-6	6.1	6.86	43	0
Lytle 7	89.1	OG	2700	W	3	214	20-35	25	4-6	7.1	11.83	45	0
East Isolation 11	89.5	MAT	2000	S	5	313	20-30	13		1.5	1.51	6	0
East Isolation 12	89.5	MIX	1800	W	5	210	15-35	16		2.4	4.9	13	0
Ennis 1	89.5	MIX	1200	N	6	202	5-30	18		3.6	19.07	19	0
Scattered 4	89.5	MAT	3600	N	5	610	20-30	19		2.6	15	17	0
Rushboro 4	89.8	MAT	3700	E	4	64,602	15-25	11		2.5	1.09	4	0
Scattered 12	89.8	MAT	3600	W	3	616	16-22	14		2.3	5.84	20	0
Ennis 3	89.9	MIX	1200	E	3	202	15-20	70		1.6	1.51	72	0
Titan Too 2	89.9	MAT	2800	E	4	23,21	10-18	12	3-4	3.0	15.29	11	5
Lookout Sentinel 5	90.2	OG	1900	E	3	231	20-25	10	3	3.4	2.44	28	0
Titan Too 4	90.2	MIX	2600	E	2	25,23	15	48	3-4	1.7	9.36	35	17
Lookout Sentinel 2&3	90.3	MAT	1700	E	3	212,16	30	25	3	2.3	3.31	22	16
French Removal 9	90.5	MAT	5000	W	3	714	20-25	21	2-3	2.2	2.29	34	0
Three Bears 1.1	90.5	MAT	3600	S	4	203	25-30	10		1.7	2.29	10	0
Three Bears 1.2	90.5	MAT	3600	W	4	203	25-30	30		1.6	15.24	20	0
Three Bears 5	90.6	MIX	3600	E	4	23	25	21		3.3	6.36	50	0
Three Bears 9	90.7	MAT	3600	E	3	233	5-13	8		2.4	0	17	0
Paws 6	90.8	MIX	2000	N	4	252	20-30	13	4-6	2.8	0.91	17	14
Roar East 6.1	90.8	MAT	3500	W	6	616	5-10	8		3.3	0	13	2
Roar East 6.2	90.8	MAT	3500	E	6	616	22	8		1.9	2.29	5	5
Roar East 7	90.8	MAT	3800	W	6	64	8-14	24		1.8	0	34	8
Stockstill 5	90.8	MIX	1600	W	2	13,162	20-30	31	4-6	2.7	2.18	66	0
Titan Too 1	90.8	MAT	2200	S	2	14	0-15	58	3-4	1.8	1.66	62	33
West Isolation 3	91.3	MAT	1800	E	5	313	15-40	12		1.0	5.13	9	0
West Isolation 1	91.3	MIX	1600	E	5	313	10-30	22		1.5	1.71	16	0
Lookout Sentinel 1	91.5	MAT	1800	N	4	212	40	10.1	5	2.6	4.45	20	0
O'Leary 6	91.5	MAT	3600	S	2	61	15-25	22	6	5.0	0.95	41	61
Sardine Boundary 3	91.5	OG	3200	E	3	214	25-30	16	4	4.8	0.65	54	17
Sardine Boundary 7	91.5	OG	4000	S	5	135	0-5	7	4	4.3	0	19	0
Slim Scout 3b	91.5	OG	2400	S	2	212	30	5	8	6.6	0	29	0
Slim Scout 4	91.5	OG	2900	N	5	210	30	12	8	9.0	2	87	0
Starrbright 6	91.5	OG	2400	E	6	231	7	8	3	4.5	4.66	15	0
Paws 1	91.6	OG	1200	N	1	15	0	22	3-4	1.8	6.83	25	0
Elk/Mink 11	91.7	MAT	4400	N	5	602	33	16	6	4.7	0	73	0
Elk/Mink 4	91.8	MAT	4000	N	5	602	30-40	23	6	6.9	1.31	148	0
Lytle 1	91.8	OG	3600	N	4	212	10-18	12	4-6	2.3	21.58	8	0
Dennis 3	92.1	MAT	1600	S	5	202	10-40	15	6	5.3	0	58	21
Dennis 1	92.2	MAT	1600	W	5	202,023	15-25	12	6	8.5	5.13	42	25
Starrbright 1	92.3	MIX	3000	E	4	14	10-25	15	4	2.8	0	36	0

* Forest type: OG=old-growth, MIX=mixed age, MAT=mature

+ Topographic position: 2=lower slope, 3=mid-slope, 4=upper slope, 5=ridge, 6=bench

Appendix 2 continued

died standing	windthrows	total % windthrw.	total % died	natural snags	fallen snags	windthrw. on edges	stand history	date topped
12	11	14.10	15.38	17	7	L		
20	18	15.52	17.24	37	9	M		
4	20	13.25	2.65	0	0	NA	thinned	91
18	42	24.56	10.53	13	11	L		
4	3	15.79	21.05	2	0	L		
0	6	37.50	0.00	2	0	M		
0	8	66.67	0.00	3	0	S		
2	0	0.00	28.57	3	0	NA		
25	29	29.90	25.77	7	5	L	blowdown	
44	78	46.71	26.35	10	12	L	blowdown	
10	1	5.88	58.82	3	0	M		
14	6	18.18	42.42	5	1	L		
5	32	57.14	8.93	8	0	S		
7	22	47.83	15.22	4	2	S		
18	1	4.35	78.26	4	0	M		
2	6	21.43	7.14	4	4	S		
8	5	5.88	9.41	30	2	L		
1	16	48.48	3.03	3	0	S	thinned	92
1	3	9.38	3.13	2	0	NA	thinned	
4	27	32.53	4.82	0	0	S	thinned	92
9	5	9.62	17.31	5	1	L	shelterwd.	90
8	3	6.67	17.78	1	1	NA	shelterwd.	
4	1	6.67	26.67	2	0	M		
8	18	39.13	17.39	3	3	M		
5	12	17.91	7.46	3	2	L		
1	0	0.00	5.56	1	0	NA		
5	1	2.70	13.51	0	1	L		91
11	0	0.00	42.31	0	0	L		
4	1	6.67	26.67	0	1	L		
0	0	0.00	0.00	0	0	NA	shelterwd.	91
7	5	6.41	8.97	5	0	L		
3	5	4.85	2.91	1	1	L	thinned	92
0	1	10.00	0.00	2	0	M		
12	1	3.45	41.38	4	0	L		
1	2	8.70	4.35	3	0	NA		
2	2	1.89	1.89	3	1	L	thinned	91
4	1	1.32	5.26	0	0	NA		92
3	0	0.00	13.64	8	0	NA	shelt/bdown	
1	0	0.00	3.33	3	0	NA		
8	4	4.04	8.08	9	0	L		
15	3	9.09	45.45	3	0	M		
8	5	13.16	21.05	2	0	L	shelterwd.	
1	0	0.00	1.35	1	0	L		
1	4	2.61	0.65	6	0	L	thinned	
8	10	38.46	30.77	2	0	S		
0	0	0.00	0.00	0	0	L	blowdown	92
5	8	10.00	6.25	22	0	S	blowdown	92
5	0	0.00	12.20	1	0	NA	thinned	

Appendix 2. Individual edge type and direction of tree-falls

						Direction windthrown trees point								
sale name	unit #	North edge	East edge	South edge	West edge	N	NE	E	SE	S	SW	W	NW	TOTAL
Dennis	1	clear-cut (cc)	cc/ridge (rdg)	forest (for)	for	5	1	1	0	0	0	0	1	8
Dennis	3	cc	for	for	cc	0	0	0	0	0	0	0	0	0
East Isolation	11	forest/ridge	for	for	for	0	1	0	0	0	0	0	0	1
East Isolation	12	for	for/rdg	for	for/river	2	0	0	2	1	0	1	0	6
East Isolation	15	cc	cc	for	cc	1	0	1	0	0	1	0	0	3
Elindel	15	for/rdg	for	for	for	0	0	6	8	0	0	0	0	14
Elk/Mink	4	for	shelterwd.	for	for	0	1	1	1	0	0	0	0	3
Elk/Mink	11	cc	for	for/rdg	for	0	0	0	0	0	0	0	0	0
Ennis	1	for/river	for	cc	cc	19	5	2	0	2	2	0	2	32
Ennis	3	for	for	for	for	0	0	1	0	1	2	1	0	5
French Removal	9	cc	cc	cc	for	0	1	0	0	1	1	0	0	3
Lookout Sentinel	2&3	for	cc/river	for	for/road	0	2	0	0	1	0	2	0	5
Lookout Sentinel	1	for	for	rdg	cc	1	0	0	0	1	0	0	0	2
Lookout Sentinel	5	for/road	for	for	cc	0	0	1	1	0	0	1	0	3
Lytle	3	cc	cc	for/river	cc	2	3	12	1	6	3	2	0	29
Lytle	7	cc/rdg	cc	for	for/river	5	0	3	5	6	5	32	22	78
Lytle	1	for	for/cc	cc	cc	0	0	6	0	0	0	3	1	10
Mendel	1	for/cc	for	for	for/river	3	1	5	0	0	0	1	1	11
Mendel	2	for/rdg	for	for/river	for/rdg	0	1	12	1	3	0	1	0	18
O'Leary	6	for	for	for/cc	for	0	1	1	0	0	0	0	0	2
Paws	1	cc	cc	road/for	for	0	1	2	0	0	1	1	0	5
Paws	6	for	cc	for	for/rdg	1	0	0	0	0	0	0	0	1
Scattered	12	for	for	for	for	1	2	0	0	1	1	1	0	6
Roar East	6	for	for	for	for	0	0	0	0	1	0	0	0	1
Roar East	7	for	for	for	for	0	0	0	0	0	0	0	0	0
Rushboro	4	for	for	shelterwd.	cc	0	0	2	0	0	0	0	0	2
Sardine Boundary	3	cc	for/river	cc	cc	0	1	0	0	0	0	0	0	1
Sardine Boundary	7	for	cc	for	cc	0	0	0	0	0	0	0	0	0
Scattered	4	for	cc	for	cc	1	11	7	2	0	0	1	0	22
Slim Scout	3b	for	for	for	for	0	0	0	0	0	0	0	0	0
Slim Scout	4	cc	for	for/rdg	for/rdg	2	1	0	0	0	1	0	0	4
Starrbright	1	cc	cc	cc	cc	0	0	0	0	0	0	0	0	0
Starrbright	6	cc	cc	for/cc	for	0	2	0	0	1	0	0	0	3
Stockstill	5	for	for	for	for	0	0	0	0	1	1	1	2	5
Three Bears	1	rdg	for	for	cc	5	8	0	0	0	2	3	1	19
Three Bears	5	for	for	for	for	0	0	5	1	4	0	1	1	12
Three Bears	9	for	for	for	for	0	0	0	0	0	0	0	0	0
Titan Too	1	cc	for	for	for/river	0	2	1	0	0	0	1	1	5
Titan Too	2	for	cc	for/cc	cc/rdg	3	9	3	0	0	1	0	0	16
Titan Too	4	for	for	for	cc	8	5	10	2	1	0	0	1	27
West Isolation	1	cc	cc/river	for	for/rdg	0	0	0	0	0	0	0	0	0
West Isolation	3	for	for	for	cc	0	0	0	0	0	0	0	1	1
Wildwood	2	for	cc	for	cc	4	10	6	0	0	0	0	0	20
Wildwood	4	for	for	road/for	cc	10	15	6	0	4	4	0	3	42