Thirty-five years of forest succession in southwest Oregon: Vegetation response to three distinct logging treatments

by
Aaron Stone Arthur

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Abstract approved:

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This study examined patterns and controls on 35-years of forest succession following logging in the 236 ha South Umpqua Experimental Forest within the Umpqua National Forest in southwestern Oregon. Prior to logging, the overstory in all three watersheds (~50% cover) was composed of Douglas-fir (30-40% cover), grand fir (2 to 10% cover), incense cedar (2-4% cover), western hemlock (1-2 % cover), sugar pine (1-2% cover), ponderosa pine (0-1% cover), and madrone (0-1% cover). The sapling cover in all three watersheds (30-35% cover) was dominated by grand fir (15-20%), western hemlock (6-8% cover) incense cedar (3-5% cover), and Douglas-fir (3-5% cover), with <1% cover by the other tree species.

In 1971, the overstory of the four watersheds received: WS-1 (69.2 ha) a 50% basal area removal (shelterwood); WS-2 (68.4 ha) 30% in 20 small clearcuts; WS-3 (49.8 ha) 100% clearcut; WS-4 (48.6 ha) uncut control. Prior to treatment, tree, shrub, and herb cover in WS 1-3 (n = 50 plots) was visually estimated using the reconnaissance survey method, but only WS 2-3 annually through 1981 (n = 39 plots). In 2006, plots were relocated and new plots added, and resurveyed (n = 77 plots). Plots were classified by environmental (aspect, slope, soil series, vegetation community) and disturbance factors

(treatment, logging method, in-watershed location, and soil disturbance). Vegetation was classed into three groups: (1) vegetation structures; (2) overstory species; and (3) residual and invader understory species.

Total vegetation cover decreased slightly over the 35-year period, with a slight decrease in shrubs, saplings, and herbs, and an increase in overstory tree cover in 2006 compared to 1971. Few statistically significant differences were observed in vegetation cover change according to environmental variables. However heavily disturbed sites were characterized by low shrub cover and high overstory cover, compared to lightly disturbed sites. These differences were still apparent 35 years after logging.

Between 1971 and 2006 in WS-2 and WS-3, Douglas fir, incense cedar and ponderosa pine cover significantly increased, while other overstory species declined. In contrast, over the same period in WS-1, there was a significant change in grand fir and western hemlock cover. Sapling cover of all species stayed constant or declined in all watersheds, except Douglas-fir and western hemlock, which increased in WS-1 only. In the post-logging period, cover of ponderosa pine and Douglas fir cover was significantly higher on moderately and heavily disturbed plots compared to lightly disturbed plots.

Before logging, residual species cover was significantly higher than invader species cover. Disturbance appeared to have the greatest effect upon the residual-invader species relationship. Residual species maintained higher cover on lightly disturbed plots, compared to invader species cover, which was highest on moderately and heavily disturbed plots. All species declined in WS-2 and WS-3 following stem exclusion.

Overall patterns of tree, shrub, and herb cover succession over the 35-year post-logging period were consistent with the expected effects of clearcutting versus thinning on light levels and their interactions with known shade tolerances of tree species.

Additionally, results were consistent with previously documented site disturbance effects on regeneration of residual and invader species, and shrub-tree competition and regeneration. These results permit prediction of the likely consequences for these 35-yr-old forest stands, and the thinned mature forest, of future, planned thinning treatments.

Thirty-five years of forest succession in southwest Oregon: Vegetation response to three distinct logging treatments

1. Introduction

Beginning in 1908, the USDA Forest Service established experimental sites to address site-specific forest, range and watershed management issues across a broad range of environmental gradients (Lugo et al. 2006). In 1951, the South Umpqua Experimental Forest totaling 1800 hectares was established primarily to study the effects of forest harvesting on hydrology and water quality within the ecotone between the cool, mesic forests of the western Cascades and the dryer, mixed-conifer forests of southwest Oregon. In 1967, the site was reduced to the four small watersheds at Coyote Creek totaling 236 hectares.

In the early 1980s, research ceased at the site, but renewed interest in 2000 resurrected the partnership between the Pacific Northwest Research Station, the Umpqua National Forest and Oregon State University to reinvigorate previous research and outline future projects at Coyote Creek. Among the historical datasets was ten years of data from permanent survey plots encompassing early secondary succession following logging initiated and collected by C.T. Dyrness and other USFS researchers.

Secondary succession studies can lend practical applications to land management concerned with timber production, invasive species, biodiversity and wildlife cover and forage. Many secondary succession studies have been based on a chronosequence approach using vegetation data collected from different sites in different successional stages (Schoonmaker and McKee 1988; Puettmann and Berger 2006). These studies can demonstrate general successional trends, but take limited account of subtle differences in microsite characters or stochastic events.

By contrast, long-term vegetation records from permanent plots can detect local factors influencing vegetation succession. For example, long-term vegetation monitoring plots at the H.J. Andrews Experimental Forest (H.J.A.) in western Cascades, Oregon have yielded invaluable data to forest ecologists. Observations from these sites revealed the resilience of vegetation communities and the emergence and decline of common species (Dyrness 1973). With species classified into two successional types, residual and invader, Dyrness (1973) illuminated the ephemeral character of many common species in post-disturbance sites and autecological requirements driving site preference. For example, the drought tolerant invader shrub, *Ceanothus velutinus*, five years following slash burning had twice the cover on burned sites than unburned sites, and it was essentially absent from the pre-logging communities (Dyrness 1973).

Observations on these permanent plots WS-1 and WS-3 in the H.J.A. continued for decades following their inception (Halpern 1988; Halpern 1989; Lutz and Halpern 2006). Using an extension of the dataset begun by Dyrness (1973), Halpern ascertained species emergence, duration and magnitude for 20 years following disturbance and noted the potential for stochastic or historic variables to influence secondary succession. For example, *C. velutinus* in WS-3 of the H.J.A. (Dyrness 1973) had similar cover on lightly and heavily burned sites, whereas it had dissimilar cover on these burn classes in WS-1 of the H.J.A. Interspecific competition, historical legacy or snow-loading were postulated as potential reasons for this discrepancy, emphasizing the advantage of monitoring permanent plots over the chronosequence approach to understanding successional pathways (Halpern 1989).

Similar to previous work at the H.J.A., the historical dataset from Coyote Creek presented here was examined to determine patterns and controls on forest succession after logging in southwest Oregon. The collected species and structure data were placed in three broad vegetation categories: vegetation structures, overstory species, and residual-invader shrubs and herbs. The percent cover of these groups was related to four environmental variables: aspect, slope, soil series, vegetation community; and four disturbance variables: logging treatment, logging method, in-treatment position and soil disturbance.

The first set of analyses examined was the three primary structural units of overstory, shrubs and herbs. These were examined to determine how environmental and disturbance variables affected the overall character and competitive interactions between the structural groups. For example, on logged sites, remnant shrubs may successfully compete with emergent saplings, shaping future stand structure (Mallick and Prescott 2001). By contrast, heavily disturbed sites with no remnant vegetation would be open to colonization by herbs and trees adapted to exposed mineral soil, but aspect and slope may affect species composition (Minore 1979).

Seven overstory tree species were the focus of the second set of analyses. Cover of these species was analyzed by landscape and treatment to determine influences on forest composition and structure in early succession. Early research on tree regeneration from Coyote Creek suggests that logging method and subsequent disturbance may have a profound influence on location of certain species within the watershed (Minore 1977). The exposure of bare mineral soil is essential for the germination of Douglas-fir, while direct light appears to be more important than soil condition for the germination and establishment of sugar pine (Minore 1979; Woodward and Land 1984; Gray et al. 2005).

The last set of analyses examined the residual and invader categories outlined by Dyrness (1973) and Halpern (1989). All shrubs and herbs were classed as either residuals or invaders to determine the interaction between remnant and introduced species, and how these groups and individuals compete (or facilitate) other structures (Oakley et al. 2006). For example, nitrogen is an important element for sapling development and a number of invader species (e.g. *Ceanothus*, legumes) have nitrogen-fixation capacities (Binkley et al. 1982). The presence or absence of specific drivers for these species' colonization may affect the density and health of subsequent colonizers.

2. Site Description

2.1 Geography

2.1.1 Location

The study site, Coyote Creek, is a 236 ha area consisting of four small northeast draining watersheds located in the South Umpqua Experimental Forest. The watersheds are located at 43°1'15"N and 122°43'30"W, approximately 65 km southeast of Roseburg, Oregon (Fig. 1). Watersheds range in area from 69.2 to 48.6 ha (Table 1).

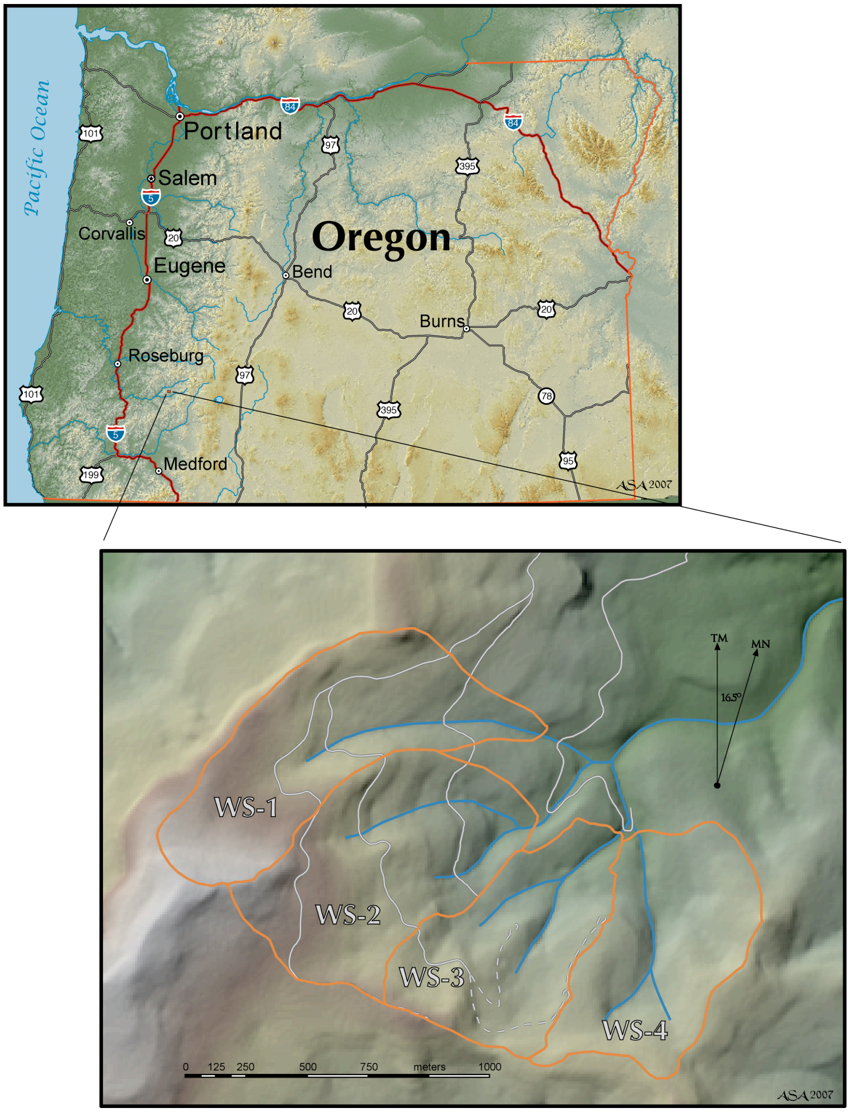

Fig. 1. Location of the South Umpqua Experimental Forest.

2.1.2 Climate

The Pacific Ocean, 150 km to the west, is the dominant climatic influence at Coyote Creek. Winters are cool and wet, and summers are warm to hot and dry. Precipitation is predominantly in the form of rain with some snowpack development in mid-winter. Approximately 90 percent of precipitation originates from Pacific cyclonic storms arriving between October and April. Mean annual precipitation over the period

1961-2003 was 125.2 cm (Harr et al. 1979; DAYMET-Thornton et al. 1997). Annual precipitation ranges from the minimum recorded annual precipitation of 87.6 cm and the maximum recorded annual of 197.7 cm (Fig. 2). Mean daily temperatures calculated over the period from 1980 to 2003 range from min -1.7°C to max 6.7°C in January and min 9.0°C to max 27.2°C in August (Thornton et al. 1997).

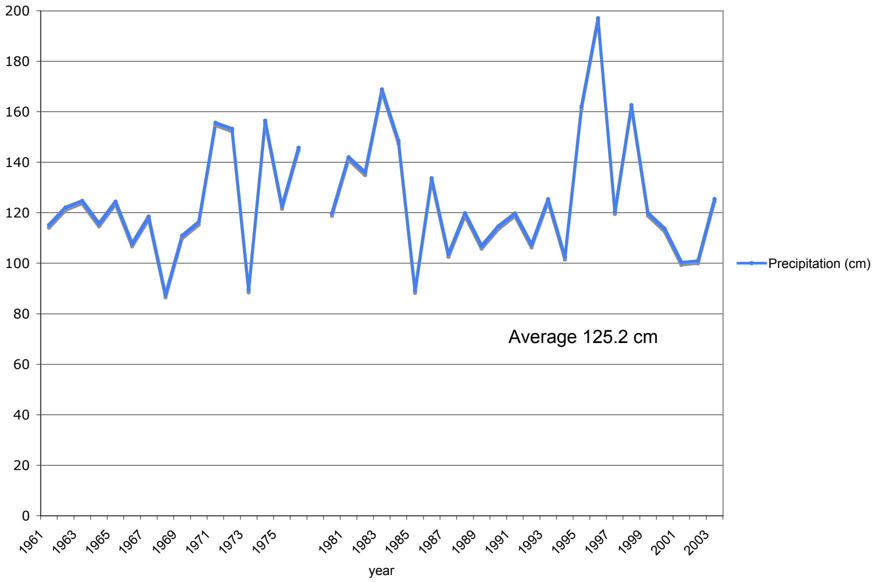

Fig. 2. Mean annual precipitation (cm) of the South Umpqua Experimental Forest, 1961-1976 (Harr et al. 1979) and 1980-2003 (derived from DAYMET-Thornton et al. 1997).

2.1.3 Geology & Soils

The Little Butte Volcanic Formation underlies Coyote Creek. This formation is composed of rhyodacitic and pyroclastic tuffs, welded-tuffs and breccias of the Miocene and Oligocene. Andesite and basalt form the ridge at Coyote Point bordering WS-4 (Kays 1970). Mass-movement sites are common with active creep, debris slides and streambank failures extensively recorded in WS-3. The dominant geomorphic process appears to be large deep-seated earthflows occurring in unwelded tuffs residing between the more resistant welded tuff layers (Swanson & Swanston 1977).

Coyote Creek lies within a Straight-Dumont soil series association area. Smaller areas within the watershed consist of the Coyata, Freezener and Zing series (Fig. 3) (Richlen 1973). Dumont soils consist of well-drained gravelly loam derived from reddish tuff and breccia parent material. They are formed from colluvium and occur on 0 to 60 percent slopes. Minimum profile depth is 150 cm with bedrock as deep as 2 meters. In the watersheds Dumont soils are found mostly on side slopes and ridges (Richlen 1973).

Straight soils consist of well-drained, very gravelly loam derived from reddish breccia. They are formed from colluvium and occur on 20 to 80 percent slopes. Profiles are shallower than the Dumont series, ranging between 50 and 100 cm. In the watersheds, these soils are found in riparian areas and on side slopes and ridges (Richlen 1973).

Coyata soils consist of shallow, well-drained gravelly loam overlaying basalt, and are found only on ridgetops in the watersheds. Deatman soils consist of shallow, well-drained gravelly sandy clay loam overlaying bedrock, are derived from greenish tuff, breccia and agglomerate, and are found on moderately to extremely steep ridges and hillsides of WS-1. Fives soils consist of shallow, well-drained loam derived from greenish tuff and breccia, and are found on gentle to steep slopes in WS-1 and WS-4. Freezener soils consist of deeper, well-drained gravelly loam derived from basalt or andesite, and are found on lower elevation side slopes. Zing soils consist of moderately drained loam derived from igneous parent material, and are found on gentle slopes in WS-1 (Richlen 1973).

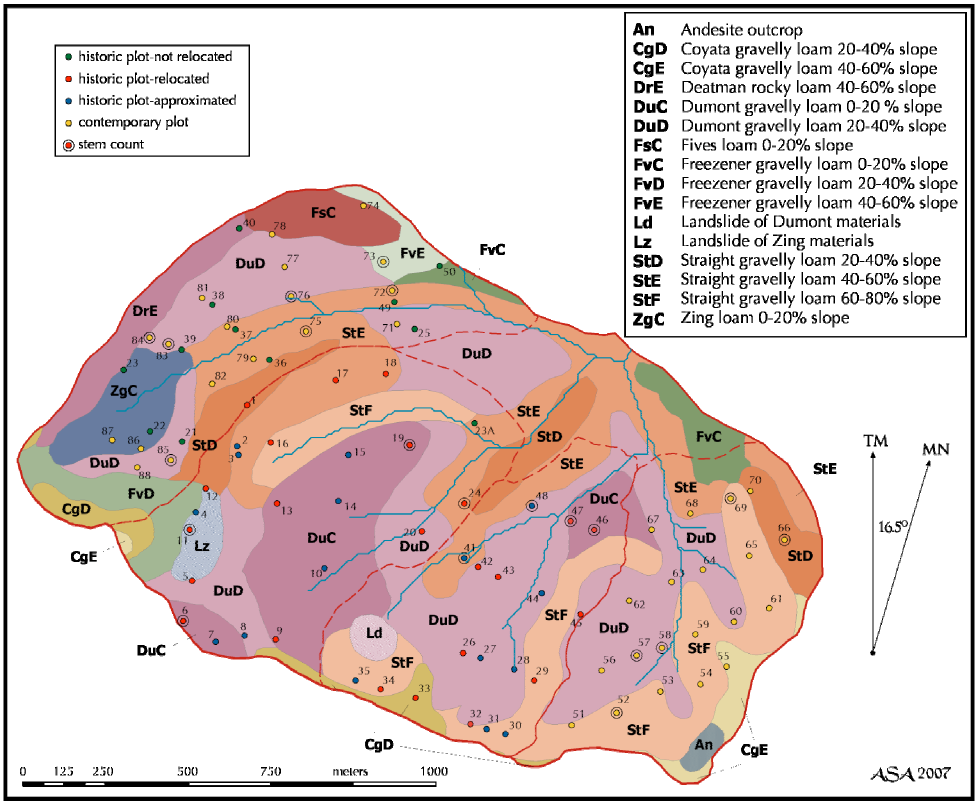

Fig. 3. Soil series of the South Umpqua Experimental Forest (after Richlen 1973).

2.1.4 Vegetation

All four watersheds lie within the mixed-conifer zone (Franklin & Dyrness 1973). Prior to logging, forests were dominated by old-growth and mature Douglas-fir (*Pseudotsuga menziesii*) with the subdominants grand fir (*Abies grandis*), sugar pine (*Pinus lambertiana*), and incense cedar (*Calocedrus decurrens*). Remaining trees of varying ages and sizes included ponderosa pine (*Pinus ponderosa*), western hemlock (*Tsuga heterophylla*) and Pacific madrone (*Arbutus menziesii*). Common subcanopy tree species included the conifer Pacific yew (*Taxus brevifolia*) and the hardwoods big-leaf maple (*Acer macrophyllum*), vine maple (*Acer circinatum*), golden chinquapin (*Castanopsis chrysophylla*), and western dogwood (*Cornus nuttallii*) (Dyrness unpublished; Minore et al. 1977).

Four canopy/understory vegetation communities were identified prior to logging (Table 3.). These communities strongly reflect a soil moisture gradient. From driest to wettest these communities were: (1) Douglas-fir/Orcutt brome-whipplea (*P. menziesii/Bromus orcuttianus-Whipplea modesta*); (2) Douglas-fir-grand fir/oceanspray (*P. menziesii-A. grandis/Holodiscus discolor-Gaultheria shallon*); (3) Douglas-fir-grand fir/salal (*P. menziesii-A. grandis/G. shallon*); and (4) Douglas-fir-western hemlock/salal (*P. menziesii-T. heterophylla/G. shallon-Linnaea borealis*) (Dyrness unpublished; Franklin & Dyrness 1973).

Very little specific data exist about succession in the South Umpqua region; however, field observations (T. Dyrness & A. Levno personal communication) and research from other sites in the western Cascades (Dyrness 1973; Halpern 1989) suggest a general successional pattern following major disturbance. Initial colonization by herbs (1-5 years) is followed by a distinct sclerophyllous shrub-field (3-5 years) primarily composed of *Ceanothus* spp and *Arctostaphylos* spp (Franklin & Dyrness 1973). In most sites *A. grandis* is the climax species with *P. menziesii* and *C. decurrens* sharing climax status on warmer, drier sites (Franklin & Dyrness 1973).

2.2 **History**

2.2.1 Management Decisions

In 1948, official interest was expressed for implementing studies on the vegetation and hydrology of southwest Oregon. In 1951, 1800 ha in the Umpqua National Forest was officially designated as the South Umpqua Experimental Forest (App. A.). The experimental forest was intended to be an outdoor laboratory to conduct experiments on single species stands, tree mortality, tree growth, and other future research projects; however, it was not to be part of the national forest on which sustained-yield was calculated. In the 1950s, the forest soils and stands were mapped (Richlen 1973). In December 1967, the experimental forest was reduced to the four small watersheds at Coyote Creek (Fig. 5).

In the 1950s, plans were drafted to install four streamflow gauges to measure daily and annual streamflow changes in each of four sub-watersheds of Coyote Creek. Streamflow volume data collection began in 1963 in all four watersheds and ran until 1981. Data collection resumed in 2001 and continues to the present. In 1967, plans were drafted to conduct a series of logging treatments on the four watersheds to monitor effects upon water quantity and quality.

The topography and stand condition of each stand influenced the prescribed logging method and treatment of each watershed in the study design. Because of gentle slopes, WS-1 was designated to receive a shelterwood cut by tractor-yarding to remove approximately one-half of the timber. For the same reason, WS-2 would receive several small clearcuts by tractor and high-lead cable system removing trees from 30 percent of the tree volume. Because it had a “thrifty” stand and more complex topography, WS-3 would be totally clearcut by high-lead cable system. Because WS-4 had the most complex topography, a poor stand and at the end of road access it would remain uncut as a control (App. A).

2.2.2. Disturbance History

In 1960, an access road and seventy-five centimeter 120-degree sharp-crested V-notch weirs were installed in WS-1 and WS-2 (Fig. 4; Table 1). In 1963, weirs of the same design were installed in WS-3 and WS-4. In summer 1970, 7.4 km of roads were constructed in WS-1, WS-2 and WS-3 for logging access. Roads occupy 1.6, 1.7 and 0.3 percent of watershed area in WS-1, WS-2 and WS-3, respectively. Twenty culverts were installed for stream crossings and to prevent overflow of ditches (Fig. 5). All roadsides were seeded, mulched and fertilized to mitigate erosion (Harr et al. 1979)

Table 1. History of logging treatments, weir construction and road-building at the South Umpqua Experimental Forest.

	WS No.			
	1	**2**	**3**	**4**
Watershed area (ha)	69.2	68.4	49.8	48.6
Elevation min-max (m)	760-1065	750-1020	730-960	730-930
Weir installation	1960	1960	1963	1963
Road area (ha)**	1.1	1.2	0.1	0
Road area (%)**	1.6	1.7	0.3	0
Pre-logging stand age (yrs)*	100-150	100-300+	150-300	100-300
Logging treatment	50% shelterwood	30% clearcut	100% clearcut	control
Year logged	1971	1971	1971	N/A
Area logged (ha)	69.2	20.5	49.8	0
Area logged (% of WS)	100	30	100	0
Area (%) logged by:				
tractor	100	14	23	0
high-lead	0	16	77	0
Area (%) slash disposed by:				
tractor	0	14	23	0
high-lead	0	16	77	0
Tree stocking (planting)***	none	7 cuts 1974	1973/1974	none
Fertilization	225kg urea-N 1972^	225kg urea-N 1970^	225kg urea-N 1971^	none

*Rothacher 1969 Study Plan

**Harr et al. 1979

***Minore et al. 1977

^Fredriksen et al. 1973

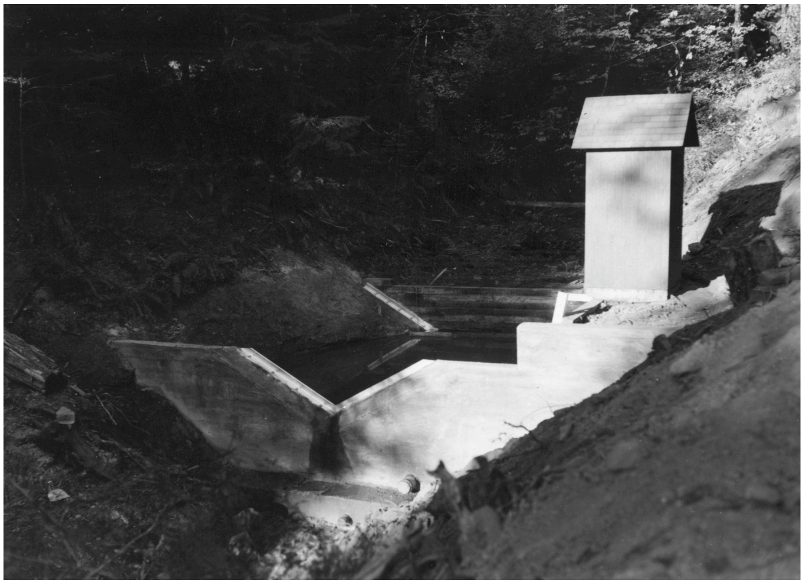

Fig. 4. 120° V-notch weir with sediment basin and gauging station in WS-3 shortly after construction, 1960 (photographer unknown).

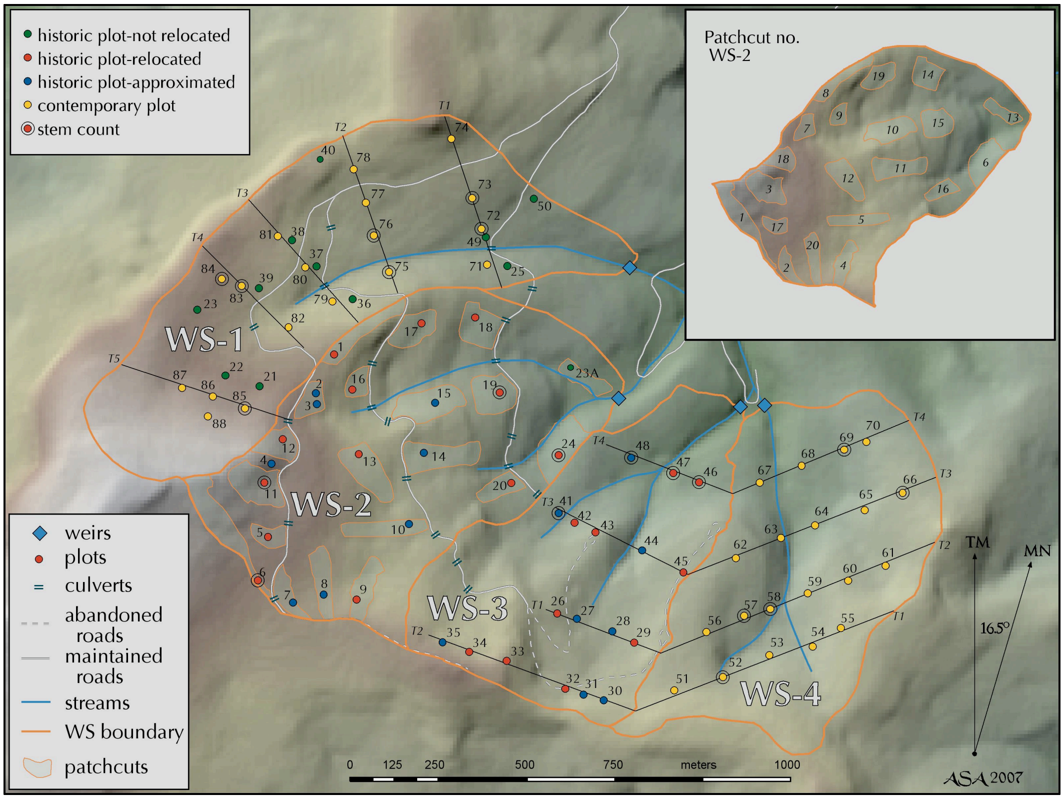

Fig. 5. South Umpqua Experimental Forest (DEM: USGS 7.5' series; Data: 1980 USFS aerial photo).

In May 1971, logging of all three watersheds began and was completed by October. Watershed 1 received a shelterwood cut totaling approximately 50 percent of the total basal area removed (Fig. 6). All logging was conducted by tractor from spur roads cut from the main road. Slash was left within the cutting area. Fertilization with 225 kg urea-N/ha was scheduled for spring 1972 (App. A).

Approximately 30 percent (20.5 of 68.4 ha) of WS-2 was clearcut in twenty small patchcuts ranging from 0.65 to 1.34 ha (Fig. 5; Table 1). Ten patchcuts on steeper sites were high-lead cable-yarded while the remaining ten on flatter sites were tractor-logged (Fig. 1). Tractor-logged sites exhibited higher soil disturbance than cable-yarded sites (Dyrness unpublished; Harr et al. 1979; Johnson and Beschta 1980). On tractor-logged sites, slash was piled and burnt; on cable-yarded sites, logs greater than 2.4 m long and 20 cm diameter were pulled to landings (Harr et al. 1979; Minore et al. 1977). Landings were ripped at the end of logging operations. Seven patchcuts were planted to Douglas-fir in spring 1974 (Minore et al. 1977). The watershed received 225 kg urea-N/ha in 1970 as part of an experiment (see Fredriksen et al. 1973; App. A).

One hundred percent (49.8 ha) of WS-3 was clearcut (Table 1). Seventy-seven percent was cleared by high-lead cable from ten landings (Fig. 6). The remaining twenty-three percent, mostly on more gentle topography, was tractor-logged. Logs greater than 2.4 m long and 20 cm diameter in cable-yarded sites were pulled to landings. Slash was dragged, piled and burned on tractor-logged sites. Landings were ripped at the end of logging operations. Planting to Douglas-fir was conducted in spring 1973 and again in 1974. Tree regeneration in the WS-2 patchcuts and WS-3 clearcut was primarily from natural seeding and planting stock in these sites was poor (Minore et al. 1977). The watershed received 225 kg urea-N/ha following logging in 1971 (App. A).

Fig. 6. Logging operations, summer 1971 (photo: USFS) (NAD 27 UTM Zone 10T).

3. Methods

3.1 <u>Field Sampling</u>

3.1.1 Sample Design

In 1970, 50 plots were located in WS-1, WS-2 and WS-3; no plots were located in unlogged WS-4 (T. Dyrness personal communication). Of the 50 plots, eleven plots were located in WS-1 using a random design; 21 plots were located in WS-2; and 18 were located in WS-3. Plots were located in WS-2 within each of the planned small patchcuts

(PC) (Fig. 5), one plot per patchcut, except for Plots 2 and 3, which were located together in PC-7, and Plot 4 located between PC-3 and PC-18. Plots were located in WS-3 along four evenly spaced northwest-southeast transects located perpendicular to the streams. The ends of the two center transects were located at landings; the upper and lower transects were spaced evenly from the center transects. Plots were spaced along transects at approximately 50 to 130 m intervals; this spacing reflected changes in the understory vegetation community or physical environment (T. Dyrness personal communication). All plots were located away from riparian areas, roads and cutting boundaries, and within sites of ecological homogeneity (T. Dyrness personal communication). All plots were marked with 3/4-inch metal conduit and hand recorded on plan-view maps (App. A).

In 2006, plots were relocated based on examination of historic maps and field notes (App. A). Each small patchcut in WS-2 was searched using two perpendicular transects (Fig. 7). In WS-3, transect end points were located and between-plot distances on the original field maps were measured. In each watershed, twenty-minute searches were conducted at each suspected plot location to relocate metal conduit markers. When historic indicators were found (soil pit, plot marker), the plot was recorded and marked. In cases where no historic indicators could be found, a new plot was located, recorded and marked. Approximately 65 percent of plots in WS-2 and WS-3 were relocated; the locations of remaining plots were approximated (Fig. 5; Table 2). Because WS-1 plots could not be relocated, and because no plots had been located in WS-4, new plots were located in both watersheds along parallel transects running perpendicular to stream channels (Fig. 5).

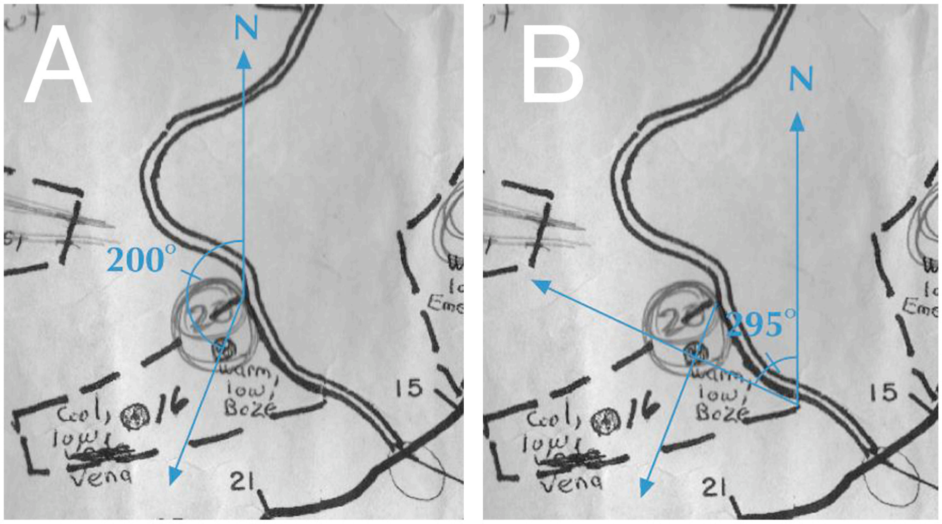

Fig. 7. Plot relocation method in WS-2. It was assumed that hand-drawn plots on historic field maps (see App. A.) were accurate relative to patchcut boundaries. Azimuth was measured from road and patchcut corner (A) and recorded in field notes. Second azimuth was measured from road and patchcut corner (B) and recorded. Search was conducted in the vicinity of azimuth crossings.

Existing metal conduit markers and/or soil pits were defined as plot centers for plots successfully relocated in WS-2 and WS-3. For replacement plots in WS-3, the new plot center was located at the measured between-plot distance based on historic maps. For replacement plots in WS-2, the new plot center was located at the intersections of perpendicular transects (Fig. 7). All original plot numbers for WS-2 and WS-3 were retained. Plots in WS-1 and WS-4 were located on transects and assigned new numbers. In WS-1 transect termini were located at road bends. In WS-4 transect termini were located at transect termini of WS-3 (Fig. 5).

Table 2. Relocation success of 39 plots in WS-2 & WS-3.

WS	**Plot**	**Found**	**Indicator**	**Method**	**WS**	**Plot**	**Found**	**Indicator**	**Method**
2	1	Y	stake	cross azimuths	3	26	Y	stake	transect distance
2	2	N	N/A	"	3	27	N	N/A	"
2	3	N	N/A	"	3	28	N	N/A	"
2	4	N	N/A	"	3	29	Y	stake/pit	"
2	5	Y	stake/pit	"	3	30	N	N/A	"
2	6	Y	stake/pit	random search	3	31	N	N/A	"
2	7	Y	pit	cross azimuths	3	32	Y	notes	"
2	8	N	N/A	"	3	33	Y	stake/pit	"
2	9	Y	pit	"	3	34	Y	stake	"
2	10	N	N/A	"	3	35	N	N/A	"
2	11	Y	stake/pit	"	3	41	Y	pit	"
2	12	Y	stake/pit	"	3	42	Y	stake	"
2	13	Y	stake/pit	"	3	43	Y	stake	"
2	14	N	N/A	"	3	44	N	N/A	"
2	15	Y	pit	"	3	45	Y	stake	"
2	16	Y	stake/pit	"	3	46	Y	stake/pit	"
2	17	Y	stake/pit	"	3	47	Y	pit	"
2	18	Y	stake/pit	"	3	48	N	N/A	"
2	19	Y	stake	"					
2	20	Y	stake	random search					
2	24	Y	stake	"					

All plots were recorded and marked using an adapted USDA Forest Service Forest Inventory and Analysis method for plot location where plot centers are recorded with GPS, marked with a stake and two witness trees marked and recorded (USDA 2006). All GPS points were recorded with a Garmin 60CSx™ hand receiver in UTM Zone 10 NAD27. Eighteen-inch cedar stakes were placed at each plot center with a steel tag denoting the plot number. Species and dbh (at 1.37 m) in cm were recorded for two witness trees within 5 meters of plot center. Each of these trees was tagged with a numbered steel tag below stump height, and the azimuth and distance in meters from the tree to the plot center were recorded. Slope, aspect and relative topographic position were recorded at plot center. All three watershed transect termini were marked with tree tags and GPS locations recorded.

Prior to logging, the vegetation and soil of each plot had been classified. Five plant communities (Table 3.) and eight soil types based on the South Umpqua Soil Survey (Richlen 1973) were identified.

Table 3. Pre-logging understory plant community in WS-1, WS-2 & WS-3. For species codes see Appendix B. (Dyrness unpublished 1971; Franklin & Dyrness 1973)

Pre-logging community		Douglas-fir/brome-whipplea	Douglas-fir-grand fir/oceanspray-salal	Douglas-fir-grand fir/salal	Douglas-fir-western hemlock/salal	Unclassified
No. plots in:						
	WS-1	5	0	5	1	0
	WS-2	7	4	7	2	1
	WS-3	0	2	1	7	8
	TOTAL	12	6	13	10	9
Moisture gradient		most xeric	⟶		most mesic	unknown
Structure						
	Shrubs	sparse	developed	developed	tall	variable
	Herbs	developed	sparse	sparse	sparse	variable
Key species						
	Trees	PSME, CADE PILA, ABGR	PSME, CADE ABGR	PSME ABGR	PSME TSHE	variable
	Shrubs	BENE, ROGY	HODI, GASH BENE, RUUR	GASH, RUUR BENE	ACCI, TABR	variable
	Herbs	FEOC, BROR LAPO, SYRE ADBI, SADO	WHMO, XETE, LIBO, POMU	WHMO, XETE LIBO, POMU	LIBO, POMU TROV, VISE	variable

3.1.2 Plot Sampling

Vegetation cover in each plot was visually estimated using a reconnaissance survey method developed on the H.J.A. Experimental Forest (Franklin et al. 1970). Vegetation was stratified into six groups: (1) overstory trees; (2) overstory tree saplings; (3) subcanopy trees and tall shrubs (average >1 meter tall); (4) low shrubs (average <1 meter tall); (5) graminoids; and (6) forbs (App. B.). Percent crown cover was estimated for total vegetation, each vegetation strata and each species within each strata. Cover was estimated to the nearest percent up to twenty percent and estimated to the nearest five percent thereafter. Trace (tr) and trace plus (tr+) were assigned for species where cover was estimated as less than 0.1 percent and 0.5 percent, respectively.

Reconnaissance survey plots are circular and contain un-standardized, nested dimensions according to vegetation layer (Franklin et al. 1970). The area assessed for total cover estimates and overstory tree estimates averaged between 15 and 20 meters in diameter. Subcanopy trees-tall shrub cover estimates were made for areas averaging between fifteen and ten meters in diameter. Low shrub, graminoid and forb cover estimates were made for areas averaging between ten and five meters in diameter (Fig. 8).

All plots in WS-1, WS-2 and WS-3 were surveyed prior to logging (Dyrness unpublished). WS-2 was surveyed in September 1970 and WS-1 and WS-3 were surveyed in April 1971. Thereafter, all plots in WS-2 (except Plot 4) and WS-3 (except Plot 44) were surveyed annually through 1979 and 1981 between May and November. After logging, in 1971, surveying in WS-1 was discontinued. In 1975, plot 23 was added to PC-13 in WS-2 (Fig. 5). In 2006, all historic plots in WS-2 and WS-3 were surveyed as well as new plots located in WS-1 and WS-4 (Fig. 5).

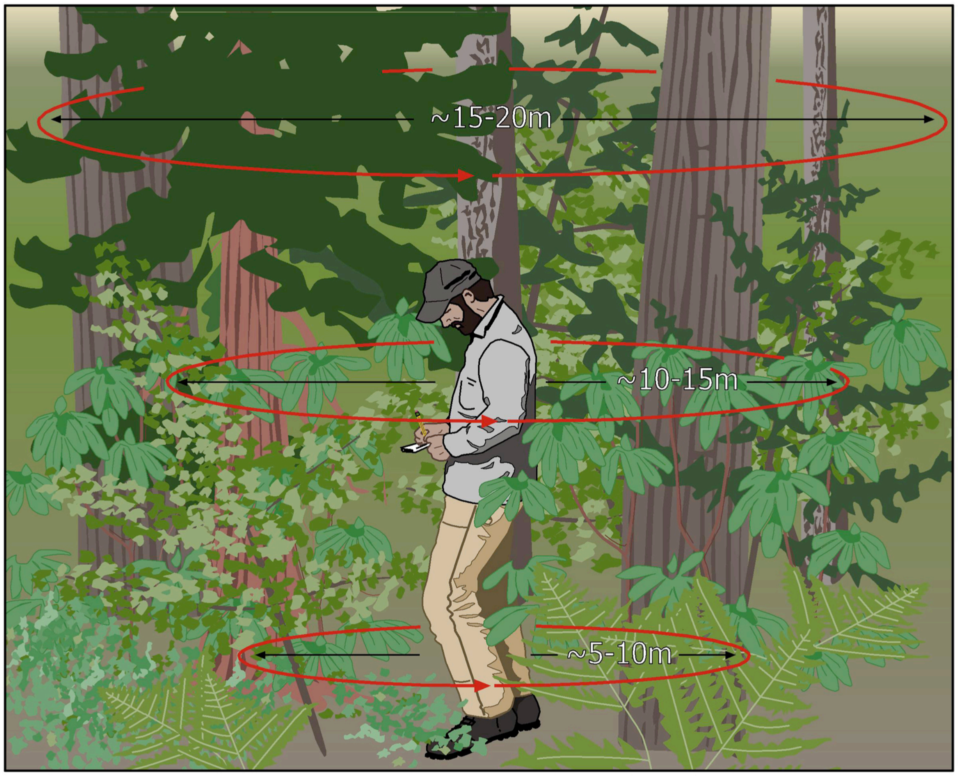

Fig. 8. Variation of survey radii for differing vegetation structures.

3.1.3 Soil Disturbance Surveys

In 1972, after logging, soil disturbance surveys were conducted to evaluate potential effects upon vegetation succession (Dyrness unpublished). Surveys were conducted at both the plot and treatment levels for WS-2 and WS-3. Both soil and slash were recorded in four classifications (Table 4). Burn disturbance was not recorded.

Table 4. Soil disturbance classifications.

Classification	Based on field observations:	WS-2	WS-3	Total
Lightly	75%+ light or undisturbed	8	6	14
Moderately	<60% heavily disturbed or compacted	7	5	12
Heavily	80%+ compacted or compacted & heavily disturbed	5	5	10
Total		20	16	36

3.1.4 Stem Counts

In 2006, to quantify tree density and species composition across the landscape, stem counts were conducted on 20 selected plots: 6, 11, 19, 24, 41 46-48, 52, 57, 58, 66, 69, 72, 73, 75, 76, 83-85 (Fig. 5). For stem count surveys, circular plots of 150m^2; 350m^2 and 615m^2 were established at existing vegetation sample plots using the same plot centers. Plots were selected because they appeared to be characteristic of the stand throughout all four watersheds based on ground observations and aerial photographs. Species, dbh in cm, height class and condition class were recorded for each tree >5 cm dbh along with azimuth and distance from plot center (App. E). No analysis of these data was conducted for this thesis.

3.2 Geographic Data Analysis

3.2.1 GIS Development

Spatial data were created and stored in ArcMap 9.2™ (ESRI 2006) for map creation, analysis and integration with the USDA Forest Service and OSU Forest Science Data Bank. All data were projected in UTM Zone 10 NAD27.

All aerial photographs were georectified on the 1980 aerial photograph (Table 5.). Additionally, roads, patchcut and watershed boundaries were based on the georectified 1980 aerial photograph. Soil data were digitized from a georectified high-resolution scan of the South Umpqua Soil Survey map (Richlen 1973).

Topographic data were based on a USGS 10m DEM. Gridded images of hillshade, aspect and slope were calculated from the DEM, as were streams and topographic lines. Vegetation survey plots and weir points were imported from ground data recorded with hand-held GPS (Table 6).

Table 5. Aerial photograph year, color, scale, source and acquisition site.

Photo Year	Color	Scale	Source	Acquired
1956	B&W	1:12,000	USFS	Knight Library UO
1971	B&W	1:15,840	"	"
1976	False IR	1:24,000	"	Tiller Ranger District
1980	B&W	1:24,000	"	Knight Library UO
1989	Color	1:12,000	"	Tiller Ranger District
1998	Color	1:12,000	"	"
2003	Color	1:12,000	"	"

Table 6. Development of GIS layers, aerial photographs and DEM.

Layer	Type	Source	Method
Stream	Polyline	USGS 10m DEM	query
Hillshade	Grid	"	"
Aspect	Polygon	"	"
Slope	"	"	"
Isohypse	Polyline	"	"
Weir	Point	GPS (2006)	inserted
Plots	"	"	"
Roads	Polyline	USFS air photo (1980)	digitize
CU Boundaries	Polygon	"	"
WS Boundaries	"	"	"
Soils	"	Richlen (1973)	"
Aerial photos	Grid	USFS (multiple years)	georectification

3.3 Data Analysis

3.3.1 Vegetation Classification and Analysis

Three vegetation classes were defined from the collected data: (1) vegetation structure, (2) overstory tree species and (3) residual and invader shrubs and herbs. Vegetation structures were the same as the vegetation stratified in field data collection (see *3.1.2 Plot Sampling*; App. B). These included mature trees and saplings of overstory species, tall and short shrubs, forbs and graminoids.

The seven overstory tree species included mature trees (>12.5 cm dbh) and saplings (<12.5 cm dbh) of Douglas-fir, grand fir, ponderosa and sugar pines, incense cedar, western hemlock and Pacific madrone. Other trees present in Coyote Creek, such as bigleaf maple and golden chinquapin, were not included in the overstory species classification because these species do not appear to maintain a regular position in the canopy in a mature stand.

Residual species included shrubs (tall and short) and herbs (graminoids and forbs) found in the stand prior to and following logging (App. B). Invader species included shrubs and herbs not present or infrequently present in the stand prior to logging, but abundant in the years following logging. When these species were found in the stand prior to logging they typically were on disturbed sites (Dyrness 1973; T. Dyrness personal communication).

Each plot was stratified into a distinct class within each of eight variables derived from environmental and disturbance characteristics. Plots were then pooled by these stratifications. Percent cover data of each vegetation class was averaged for each pooled plot stratification and mean averages compared between them. Two-sample *t*-tests were applied to comparisons between two stratifications, while ANOVA with a Tukey's highest standard deviation test was applied to comparisons between three or more stratifications. Microsoft Excel™ and the statistics software S-Plus 6.1™ were used for all statistical comparisons.

3.3.2 Plot Stratification: Environmental Variables

Plots were stratified by the environmental variables of aspect, slope, soil series and vegetation community. It was assumed that aspect was indicative of solar radiation; southerly-facing slopes have higher solar energy than northerly-facing slopes. To test the potential effects of solar radiation on vegetation succession, only plots in WS-2 and WS-3 were grouped within the eight cardinal directions (Figs. 9 and 10; Table 7). There were no south-, southwest- or west-facing plots present for sampling. Aspects of plots in WS-2 and WS-3 were measured in the field in 2006. Because aspect was not measured in the field in 1971, WS-1 was not included in the analysis.

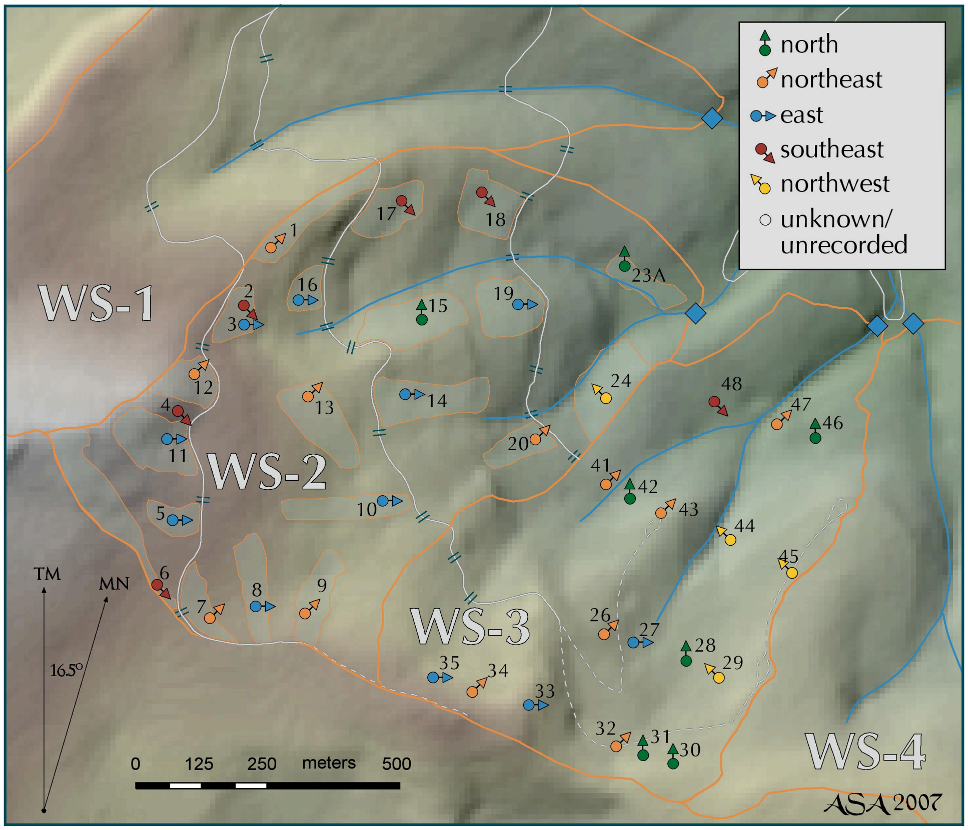

Fig. 9. Aspect stratification in WS-2 & WS-3.

Table 7. Aspect stratification and number of plots per watershed.

Aspect degrees	Quarter direction	WS-2	WS-3	TOTAL
337.5 - 22.5	North	1	5	6
22.5 - 67.5	Northeast	6	6	12
67.5 - 112.5	East	8	3	11
112.5 - 157.5	Southeast	5	1	6
292.5 - 337.5	Northwest	1	3	4
	TOTAL	21	18	39

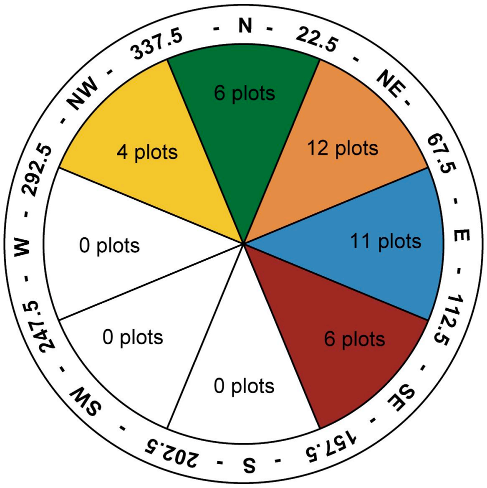

Fig. 10. Compass with number of plots per quarter aspect in degrees.

It was assumed that slope gradient was indicative of soil moisture availability, because steep slopes tend to contain shallow, stony soils. To test the effects of slope on succession, only plots in WS-2 and WS-3 were stratified into one of three slope classifications: gradual, moderate or steep (Table 8; Fig. 11). Plots from WS-1 were not included in analysis because exact location of pre-logging plots was unknown. Slope percentages were measured in the field using a Suunto PM-5™ clinometer in 2006. Because slope percentages were not measured in the field in 1971, WS-1 was not included in the analysis.

Table 8. Slope stratification and number of plots per watershed.

Slope percent	Slope class	No. of plots WS-2	WS-3	Total
< 20	Gradual	8	8	16
20-40	Moderate	8	6	14
> 40	Steep	3	5	8

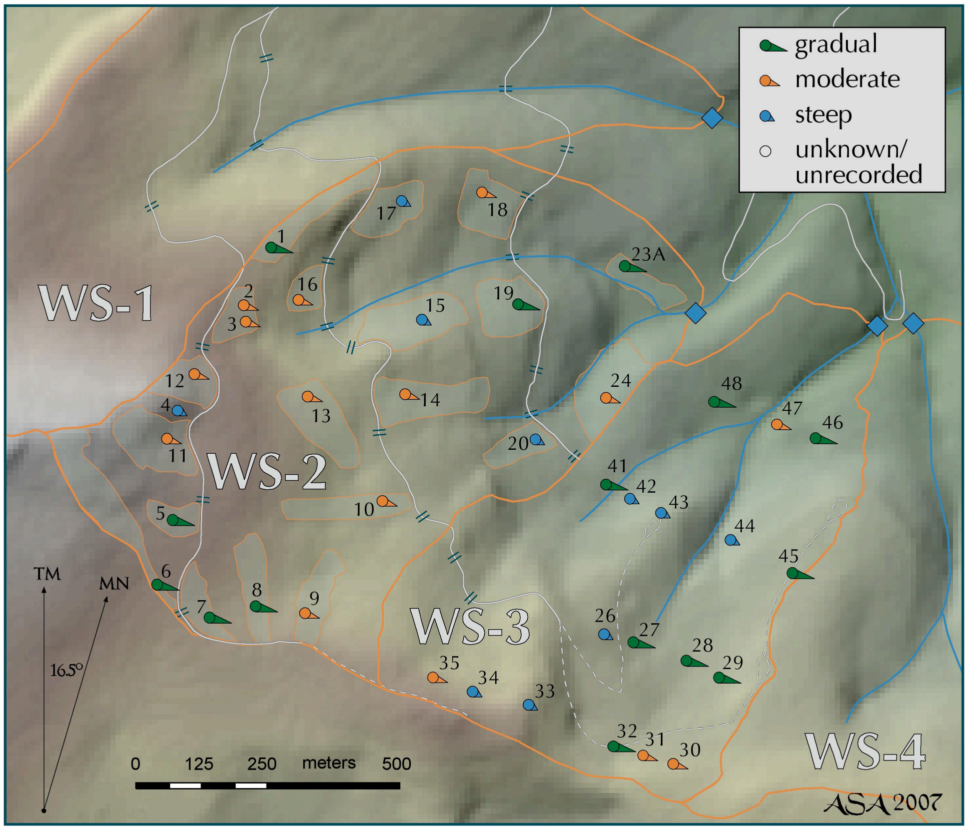

Fig. 11. Slope stratification in WS-2 & WS-3.

Despite the similarity between the soils present in the site, plots were stratified by soil series using the South Umpqua Soil Survey (Fig. 3). Only plots in WS-2 and WS-3 were stratified by soil series. Twenty-two plots were on Dumont soils, 13 plots were on Straight soils and the remaining three plots were classed as other (1 Coyata and 2 Zing soils).

It was assumed that pre-logging vegetation communities were indicative of soil moisture and solar radiation availability. To test the resiliency of vegetation communities, plots were stratified by vegetation community (Table 3; Fig. 12). Because vegetation communities were not determined for relocated plots in WS-1 in 2006, only plots from WS-2 and WS-3 were stratified and analyzed by vegetation community.

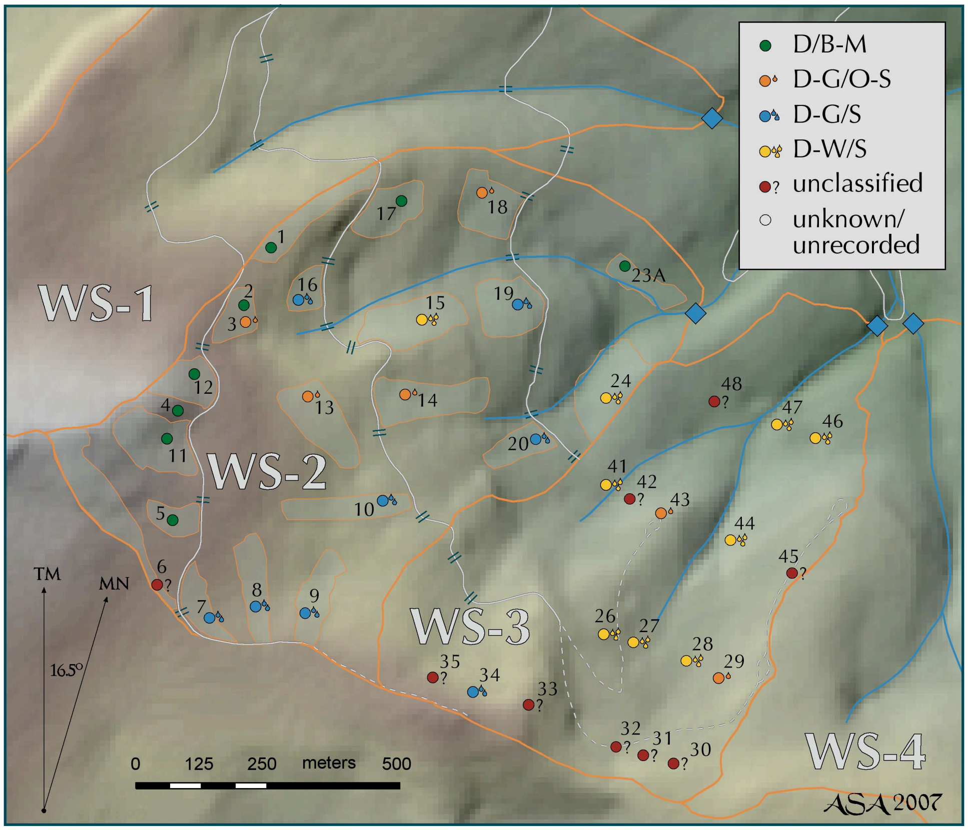

Fig. 12. Vegetation community stratification in WS-2 & WS-3. D/B-M = Douglas-fir/brome-whipplea; D-G/O-S = Douglas-fir-grand fir/oceanspray-salal; D-G/S = Douglas-fir-grand fir/salal; Douglas-fir-western hemlock/salal

3.3.3 Plot Stratification: Disturbance Variables

Plots were stratified by the disturbance variables of logging treatment, logging method, edge to center location and soil disturbance. Plots in WS-1, WS-2 and WS-3 were stratified by logging treatment and compared. Because WS-1 was not continually surveyed, all three watersheds were compared for 1971 and 2006 only. Plots in WS-2 and WS-3 were compared for the entire period of observation (1971-1979, 1981, 2006).

Of the twenty patchcuts in WS-2, half were logged by tractor and half were logged by high-lead cable system (Fig. 5; Fig. 6). Plots in WS-2 were stratified by logging method and analyzed to determine effects upon succession. Plots in WS-3 were not included in the analysis because it was not possible to determine the exact logging method for every plot. Plots in WS-1 were not included in the analysis because they were not clearcut.

Plots in WS-3 were classified as either edge or center plots to observe any differences in succession, particularly in the ratio of residual and invader species (Fig. 13). Edge plots were all within 50 m of the watershed boundary. Plots in WS-2 were not included in the analysis because the patchcuts would not be large enough to determine edge effects in succession. Plots in WS-1 were not included in the analysis because of the difference in treatment (shelterwood cut).

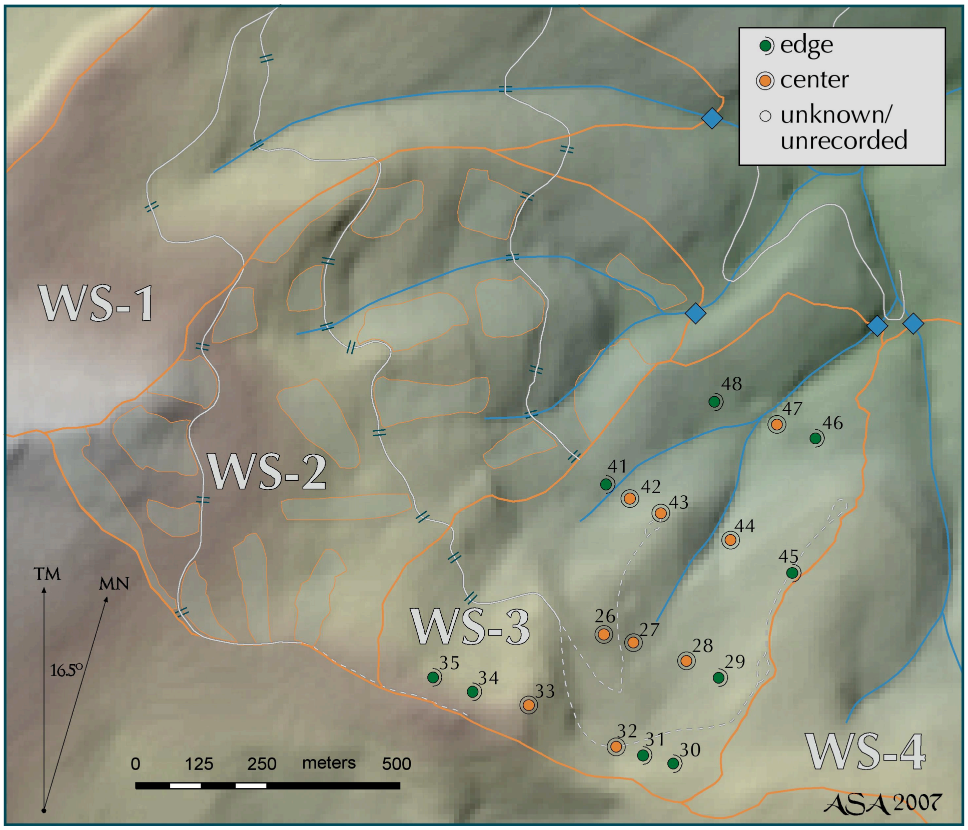

Fig. 13. Relative location (edge or center) stratification in WS-3.

It was assumed that severity of soil disturbance affects individual species independently; therefore, vegetation groupings would be affected independently as well. Only plots in WS-2 and WS-3 were stratified by the severity of soil disturbance (Table 4; Fig. 14). Plots in WS-1 were not included in the soil disturbance analysis because soil disturbance surveys were not conducted in the watershed following logging.

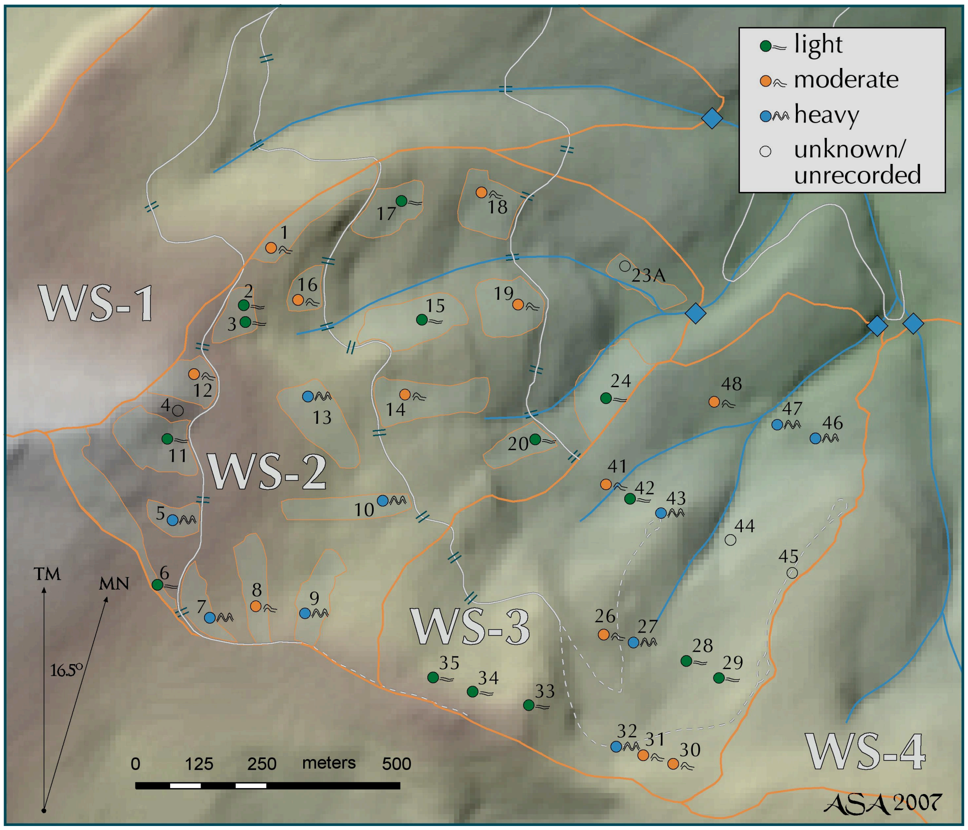

Fig. 14. Soil disturbance stratification in WS-2 & WS-3 (Dyrness unpublished 1972).

4. Results

4.1 Change in Vegetation Structure

4.1.1 Effect of Shelterwood Harvesting, Clearcutting and Small Patchcutting on Vegetation Cover

In the pre-logging period, total cover in WS-2 and WS-3 averaged 100 percent (Fig. 15). Overstory canopy cover and shrub cover were roughly equal (52.1 and 56.5 percent respectively), while herbs had the lowest cover in the pre-logging period (28.8 percent) (Fig. 15).

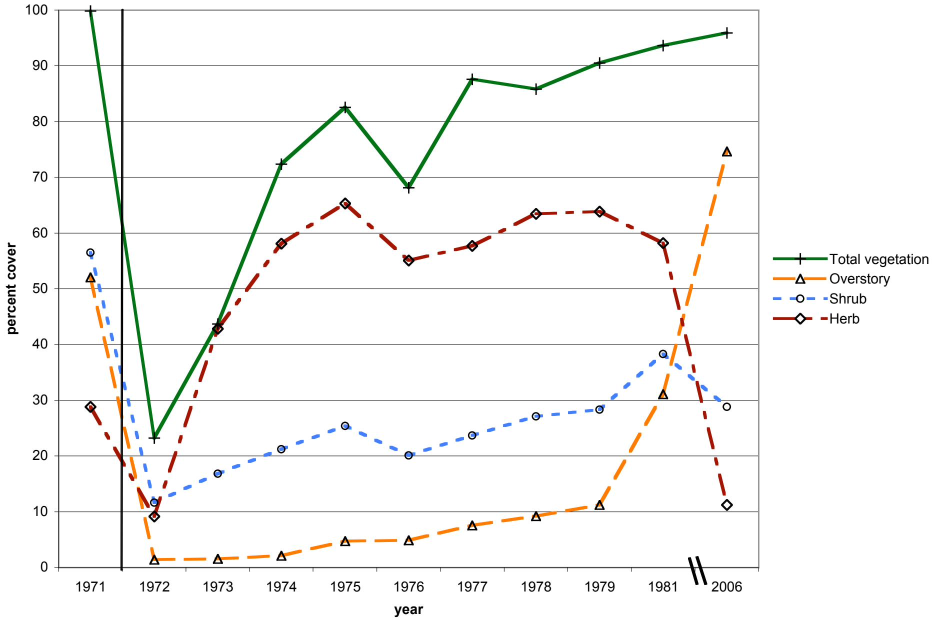

Fig. 15. Change in total vegetation, overstory canopy, shrub & herb cover in WS-2 & WS-3 (n = 39 plots).

In WS-2 and WS-3, cover of all vegetation components increased immediately following the post-logging decline; however, cover of all components declined five to six years following logging. Total vegetation cover and overstory tree cover continued to increase, while shrub and herb cover declined between ten and thirty-five years post-logging. Cover of short shrubs (<1 m) was higher than cover of tall shrub (>1 m) throughout the period of observation (Fig. 16). Forb cover was higher than grass cover throughout the period of observation; however, forb cover declined after the fourth year post-logging, while grass cover increased in the same year (Fig. 17).

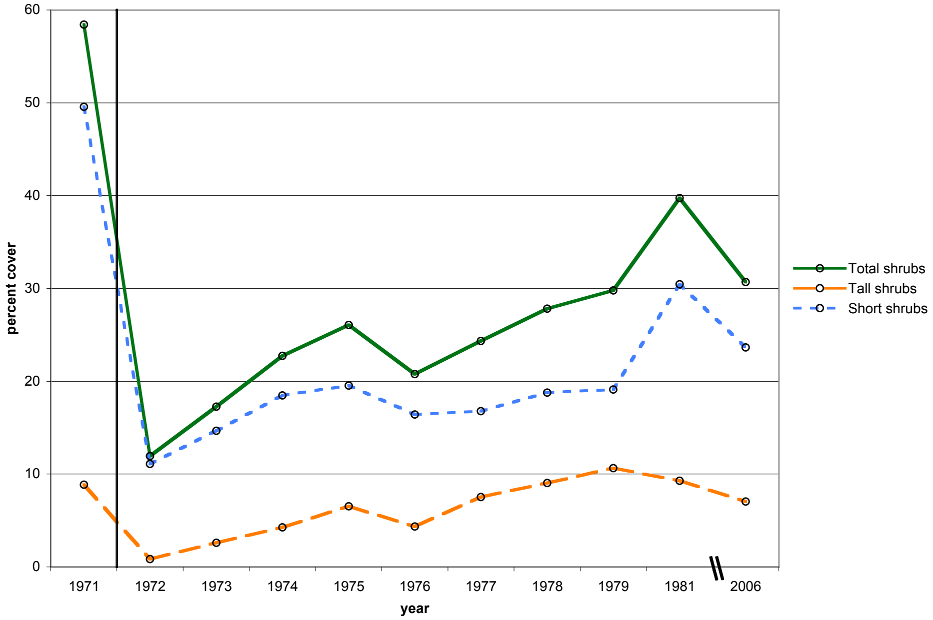

Fig. 16. Change in tall, short and total shrub cover in WS-2 & WS-3. Shrub total sum of tall + short shrubs (n =39 plots).

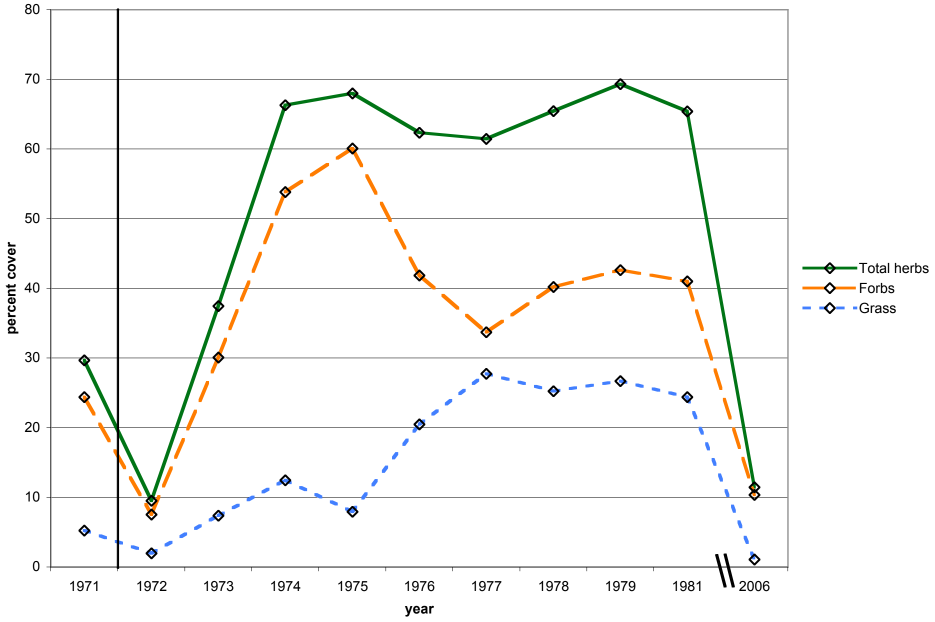

Fig. 17. Change in grass, forbs and total herb cover in WS-2 & WS-3. Herb total sum of grass and forbs (n = 39 plots)

Considering all three treated watersheds (1971: n = 50 plots; 2006: n = 57 plots), total vegetation cover in 2006 was slightly less than in 1971; saplings, short shrubs and forbs had 10 to 20 percent less cover in the 35-year old post treatment forest of 2006 compared to the pre-logging forest of 1971. However, overstory cover of trees in 2006 was approximately 20 percent greater than in the pre-logging forest of 1971 (Fig. 18).

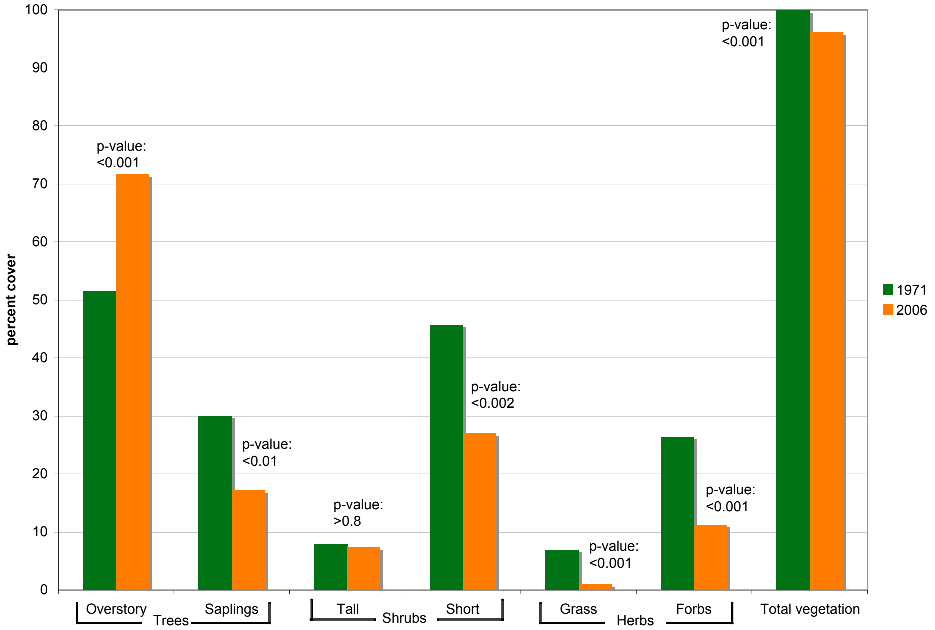

Fig. 18. Change in structural unit cover between 1971 (pre-logging) & 2006. WS 1-3 (1971: n = 50 plots; 2006: n = 57 plots) (p-values from two sample *t*-test).

4.1.2 Relationship between Environmental Characteristics and Succession

__Relationship of aspect to vegetation succession after patchcutting (WS-2) and clearcutting (WS-3)__

Prior to logging, all WS-2 and WS-3 plots stratified by aspect had roughly equivalent total vegetation cover (99.5 to 100 percent) (Fig. 19). Immediately following logging, total vegetation cover declined among these plots irrespective of aspect. One year following logging, total vegetation cover was highest on northwest-facing plots (26.8 percent) and lowest on northeast-facing plots (18.6 percent); however, this difference was not statistically significant. Thirty-five years post-logging, total vegetation cover was roughly equivalent regardless of aspect (94.4 to 98.5 percent).

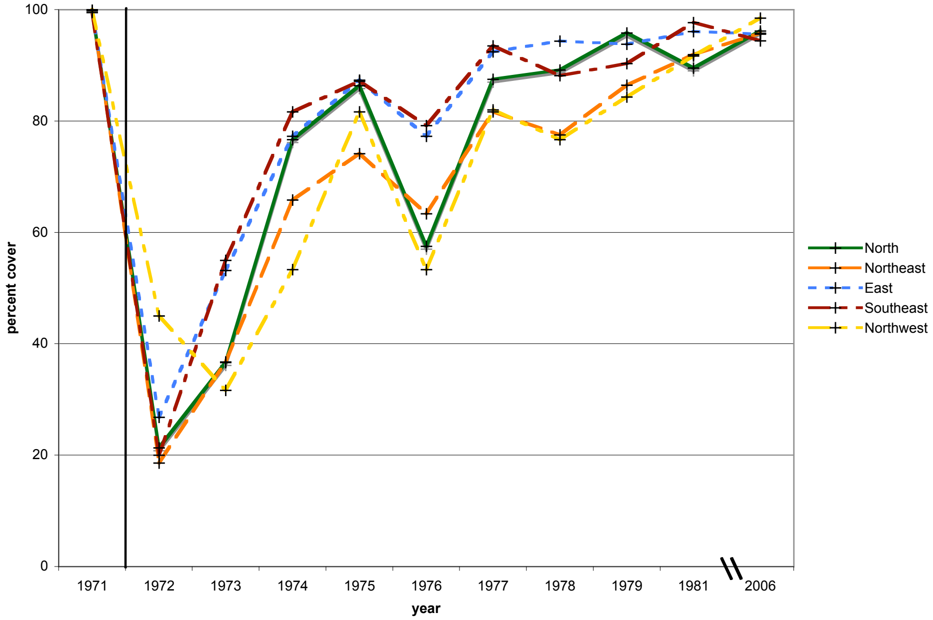

Fig. 19. Change in total vegetation cover in WS-2 & WS-3; plots stratified by aspect (n = 39 plots).

Prior to logging, overstory canopy cover was highest on north-facing plots (60 percent) and lowest on northwest-facing plots (40 percent); however this difference was not significant. Immediately following logging, overstory cover was highest on northwest-facing plots (12.6 percent), while all other plots had much lower cover (0.7 to 1.1 percent). Overstory cover was highest on northwest-facing plots from one to ten years post-logging until the thirty-fifth year when cover values were overtaken by northeast- and north-facing plots.

Prior to logging, shrub cover was highest on northwest-facing plots (71.8 percent) and lowest on southeast-facing plots (21.5 percent). In the years following logging, shrub cover did not exhibit clear dominance or statistical difference on any aspect. Thirty-five years post-logging, the highest shrub cover was on northwest-facing plots (59.8 percent) and lowest on east- and southeast-facing plots (17.6 percent respectively).

In the pre-logging period, herb cover was highest on southeast-facing plots (39.5 percent) and lowest on east-facing plots (22.3 percent); however, this difference was not significant (Fig. 20). Throughout the first ten years post-logging, the highest herb cover was consistently on southeast- and east-facing plots and the lowest cover on northwest-facing plots. In the third, seventh and tenth years post-logging, herb cover on southeast-facing plots was significantly higher than on northwest-facing plots. Thirty-five years post-logging, the highest cover continued to be on southeast-facing plots (21.4 percent) and lowest on northeast-facing plots (7 percent); however, this difference was not statistically significant.

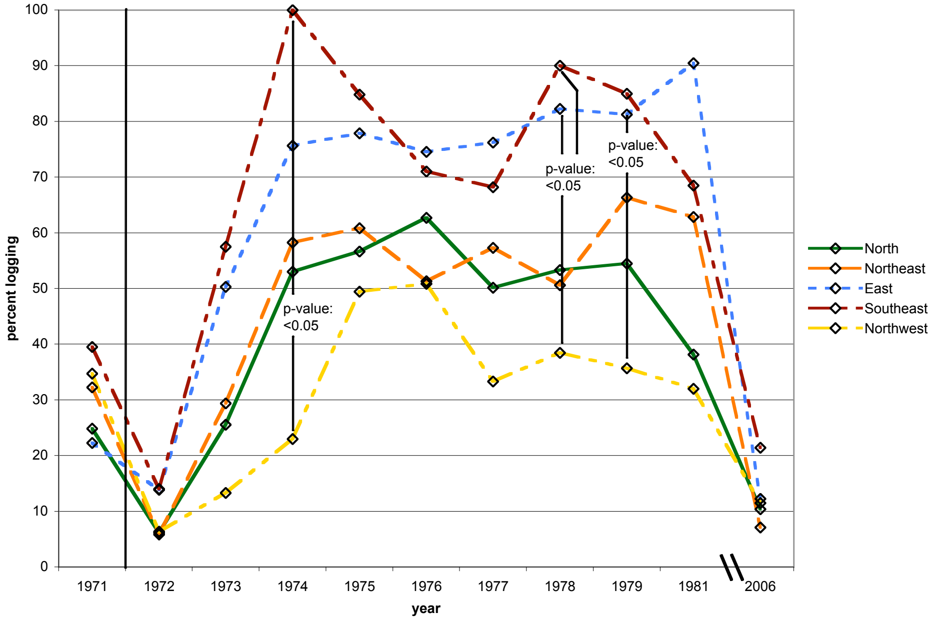

Fig. 20. Change in herb (grass + forbs) cover in WS-2 & WS-3; plots stratified by aspect (n = 39 plots) (p-values from ANOVA)

Relationship of slope to vegetation succession after patchcutting (WS-2) and clearcutting (WS-3)

In the pre-logging period, total vegetation was similar (99.5 to 100 percent) on all three slope classes (gradual, moderate and steep); however, overstory cover was highest on steep slopes, shrub cover was highest on steep and gradual slopes and herb cover highest on gradual slopes. Only the difference in herb cover was significant (gradual and steep slopes; ANOVA, p-value <0.05).

Throughout the first through tenth years post-logging, total vegetation and herb cover values were consistently highest on moderate slopes. Overstory cover was significantly higher on gradual slopes than steep slopes in years seven, eight and ten post-logging (Fig. 21). Shrub cover was significantly higher on steep slopes than gradual and moderate slopes in years three, ten and thirty-five years post-logging (Fig. 22). Overall, overstory cover was highest on gradual slopes and lowest on steep slopes, while shrub cover was highest on steep slopes and lowest on gradual slopes.

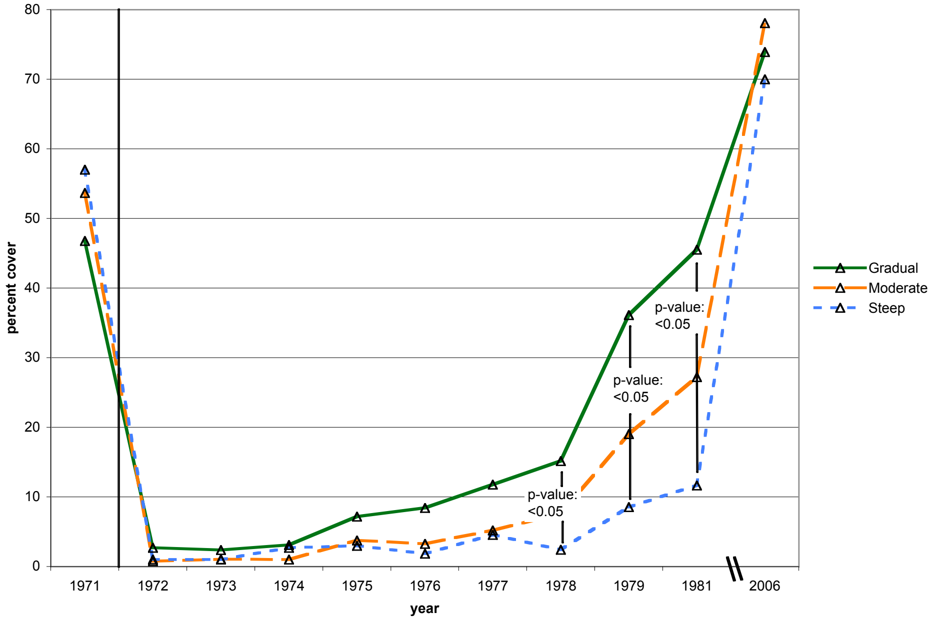

Fig. 21. Change in overstory canopy cover in WS-2 & WS-3; plots stratified by slope (n = 39 plots) (p-values from ANOVA)

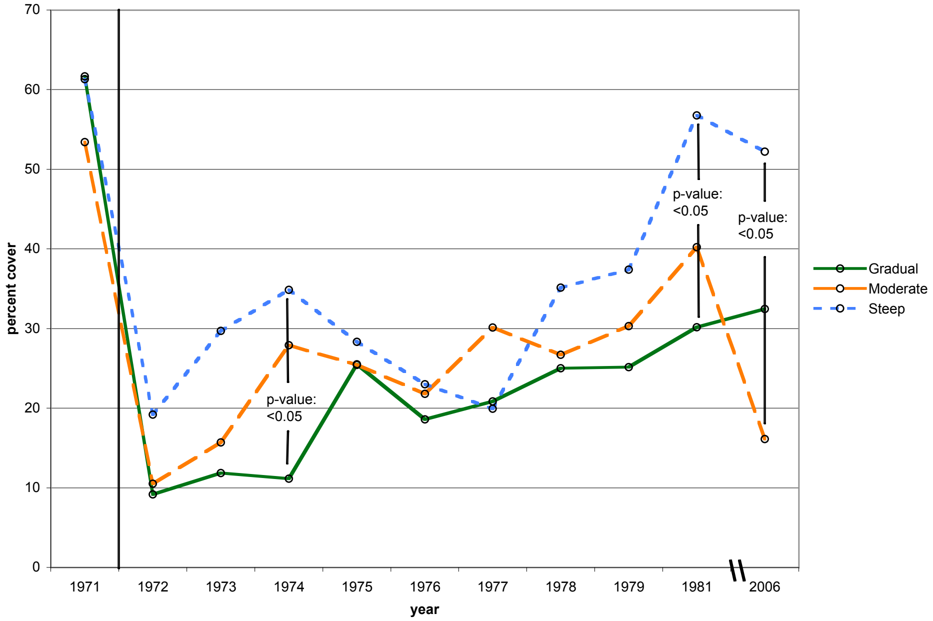

Fig. 22. Change in (tall + short) shrub cover in WS-2 & WS-3; plots stratified by slope (n = 39 plots) (p-values from ANOVA)

Relationship of soil series to vegetation succession after patchcutting (WS-2) and clearcutting (WS-3)

In the pre-logging period, total vegetation cover was similar on all soil types (~100 percent). Overstory canopy cover was highest on other soil types (Coyata and Zing soils; 66.3 percent), but approximately equal on Straight and Dumont series (52.3 and 49.3 percent respectively). Shrub cover was significantly higher on Dumont and Straight soils (71.9 and 48.7 percent, respectively) than on other soils (15.8 percent) (Fig. 23). Herb cover was similar on all soil types (28.6 to 33.5 percent).

Throughout the post-logging period, total vegetation cover and overstory cover were insignificantly higher on Straight and other soils than Dumont soils. Shrub and herb cover were not consistently higher on any soil type. Herb cover was significantly higher on other soils (100 percent) than Straight soils (52.4 percent) in the tenth year; however, this was the only significant difference in the post-logging period of observation (ANOVA, p-value <0.05).

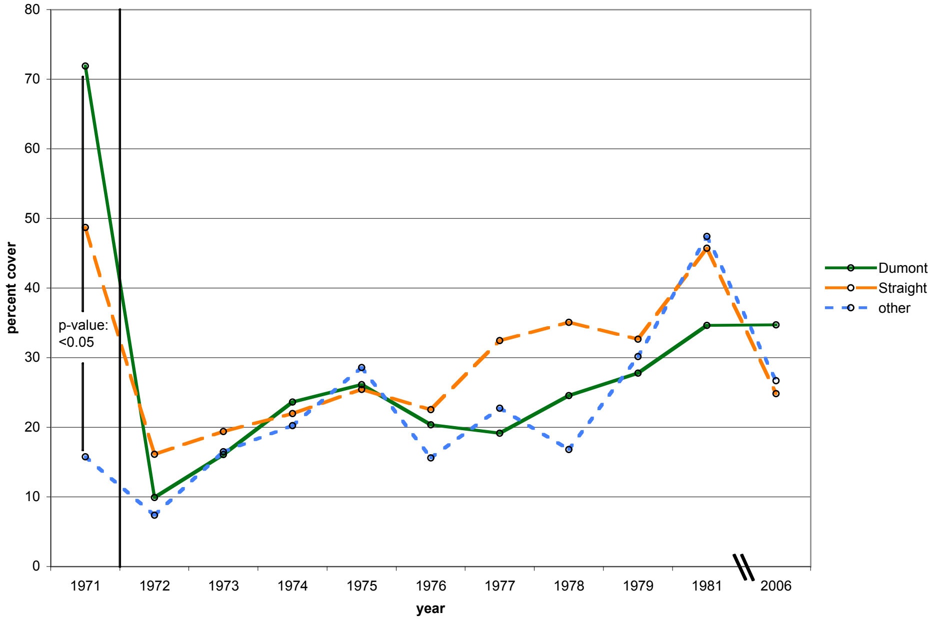

Fig. 23. Change in (tall + short) shrub cover in WS-2 & WS-3; plots stratified by soil series (n = 39 plots) (p-values from ANOVA)

Relationship of vegetation community to vegetation succession after patchcutting (WS-2) and clearcutting (WS-3)

In the pre-logging period, total vegetation cover was similar in all vegetation communities (~100 percent). Overstory cover was highest in the Douglas-fir/brome-whipplea community (60 percent) and lowest in the Douglas-fir-grand fir/oceanspray-salal (44.2 percent). Throughout most of the post-logging period, total vegetation cover was highest on Douglas-fir/brome-whipplea and Douglas-fir-grand fir/salal communities and lowest on unclassified community plots. Overstory canopy cover was highest on Douglas-fir/brome-whipplea community plots for the majority of the post-logging period. At no point were the total vegetation and overstory canopy cover differences statistically significant.

In the pre-logging period, shrub cover was significantly higher on Douglas-fir-grand fir/oceanspray-salal, Douglas-fir-grand fir/salal and Douglas-fir-western hemlock/salal communities (88.8, 87.3 and 80 percent respectively) than both Douglas-fir/brome-whipplea and unclassified communities (9.5 and 28.9 percent respectively) (Fig. 24). Throughout the majority of the post-logging period, shrub cover was highest in the Douglas-fir-grand fir/oceanspray-salal community and next highest in Douglas-fir-grand fir/salal and Douglas-fir-western hemlock/salal communities. However, at no point in the post-logging period were the differences statistically significant.

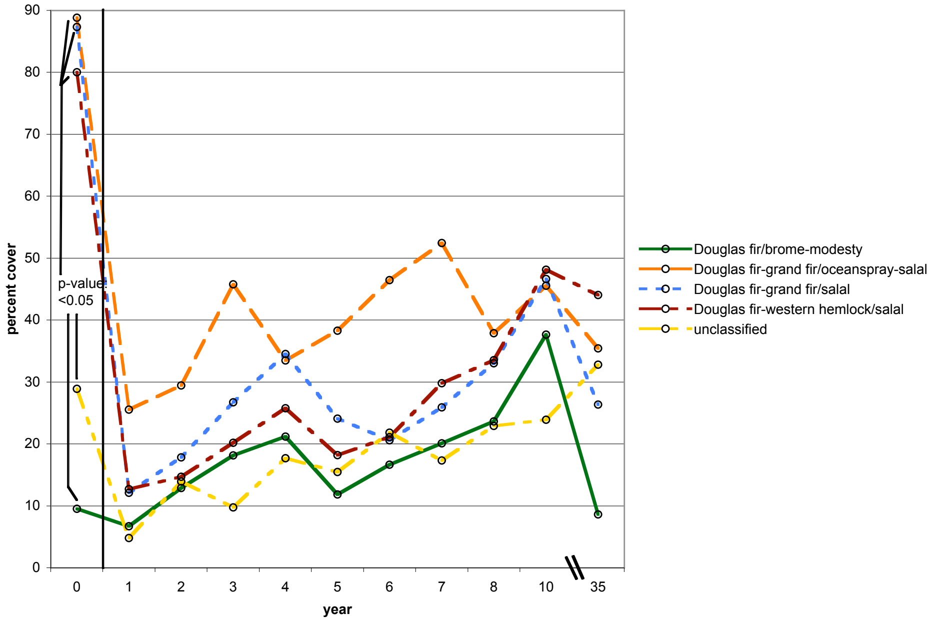

Fig. 24. Change in (tall + short) shrubs in WS-2 & WS-3; plots stratified by vegetation community (p-values from ANOVA)

Herb cover was highest in the Douglas-fir/brome-whipplea community (46.7 percent) and low in all other communities (23.9 to 28 percent); however, these differences were not significant. In years one through four and seven, herb cover was highest in the Douglas-fir/brome-whipplea community. In years eight, ten and thirty-five, herb cover was highest in the Douglas-fir-grand fir/oceanspray-salal community.

4.1.3 Relationship between Disturbance and Succession

Relationship of logging treatment to vegetation succession after shelterwood thinning (WS-1), patchcutting (WS-2) and clearcutting (WS-3)

In the pre-logging period, total vegetation cover was similar in WS-2 and WS-3 (~100 percent); however, overstory, shrub and herb cover were all insignificantly higher in WS-2 than WS-3. Overstory canopy cover did not differ between patchcuts in WS-2 and the clearcut in WS-3 at any time throughout the post-logging period. Although shrub cover was consistently higher in WS-2, only during three and thirty-five years post-logging was the difference statistically significant (two sample *t*-test; p-value <0.04 and <0.03 respectively).

Total vegetation cover and herb cover were significantly higher in the WS-2 patchcuts than the WS-3 clearcut throughout much of the post-logging period. In years two, five through eight, ten and thirty-five post-logging, total vegetation cover was significantly higher in the patchcuts of WS-2 than in the WS-3 clearcut (Fig. 25). Total vegetation cover declined in both watersheds between the fourth and fifth years post-logging; however, only the decline in WS-3 was significant.

Herb cover was significantly higher in the WS-2 patchcuts than in the WS-3 clearcut during the second, third, and fifth-seventh years post-logging (Fig. 26). Herb cover declined in WS-2 between the third and fourth years and declined in WS-3 between the fourth and sixth years post-logging; however, only the decline in WS-3 was significant.

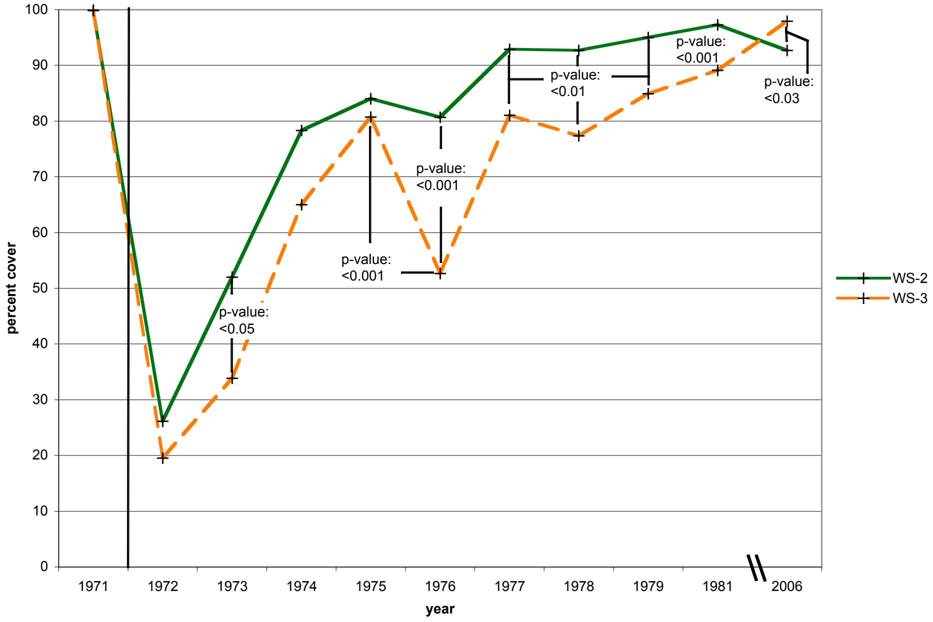

Fig. 25. Change in total vegetation cover in WS-2 & WS-3; plots stratified by treatment (n = 39 plots) (p-values from two sample *t*-test)

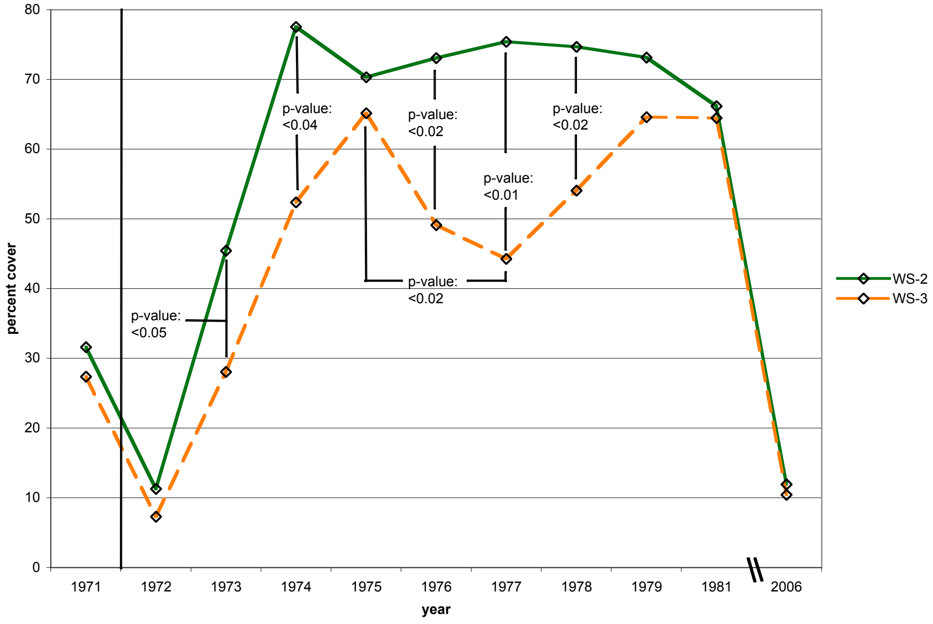

Fig. 26. Change in herb (grass + forb) cover in WS-2 & WS-3; plots stratified by treatment (n = 39 plots) (p-values from two sample *t*-tests)

Considering all three treated watersheds (1971: n = 50 plots; 2006: n = 57 plots) and the control (2006: n = 77 plots), mature overstory cover in 2006 was higher than in

1971 (Figs. 27 and 28). Total vegetation, grass and forbs had 1 to 20 percent less cover in the 35-year old post-treatment forest of 2006 compared to the mature forest of 1971. Overstory saplings and short shrubs had higher cover in 2006 than 1971 in WS-1 only. Tall shrubs had higher cover in 2006 than 1971 in both WS-1 and WS-2.

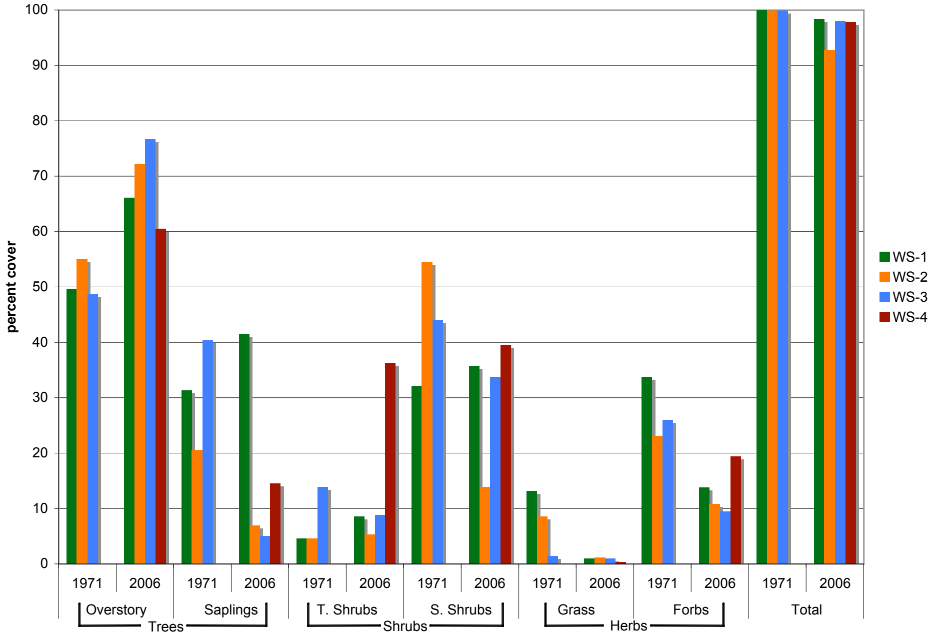

Fig. 27. Total cover in 1976 & 2006 of each structural unit of WS 1-4; plots stratified by treatment (1971: n = 50 plots; 2006: n = 57 plots (n = 77 including WS-4)).

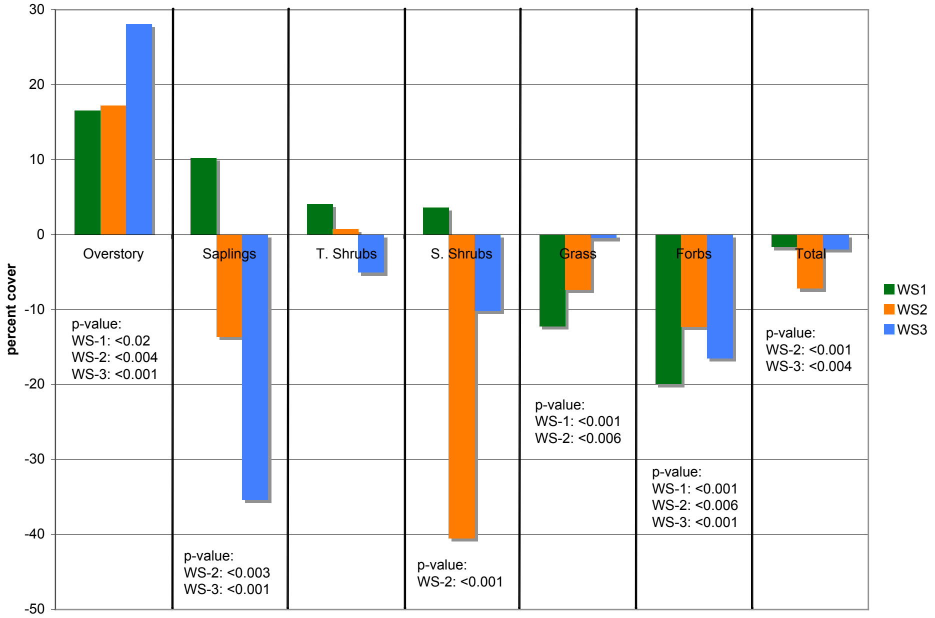

Fig. 28. Change in structural units between 1976 & 2006 in WS 1-3 (1971: n = 50 plots; 2006: n = 57 plots) (p-values from two sample *t*-test).

Relationship of in-watershed location to vegetation succession after clearcutting (WS-3)

In the pre-logging period, total vegetation (99.7 and 100 percent), overstory canopy (54 and 53.7 percent) and shrub (44.5 and 45.1 percent) cover were roughly equal between edge and center plots. Herb cover was insignificantly higher on edge plots (33.8 percent) than center plots (20.8 percent).

Throughout the post-logging period, total vegetation and overstory cover were higher on edge plots than center plots. In the first, second, tenth and thirty-fifth year herb cover was higher on center plots than edge plots. In the thirty-fifth year, shrub cover was higher on center plots than edge plots. Only in the tenth year post-logging did total vegetation have significantly higher cover on edge plots (93.8 percent) than center plots (83.7 percent; two sample *t*-test, p-value <0.004). Thirty-five years after logging, overstory cover was significantly higher on edge plots than center plots (97.2 and 76.7 percent respectively; two sample *t*-test, p-value <0.05).

Relationship of logging method to vegetation succession after patchcutting (WS-2)

In the pre-logging period, cover of total vegetation (100 and 99.8 percent) and overstory (53.3 and 56.3 percent) was similar on plots that would be tractor-logged to those that would be cable-yarded. Shrub cover was higher on the plots that would be cable-yarded plots than the plots that would be tractor-logged (54.8 and 62.2 percent respectively), while the inverse was true for herb cover (40.2 and 25.2 percent respectively); however, neither difference was statistically significant.

Total vegetation cover was higher in cable-yarded plots than tractor-logged plots in the first five years following logging, but was roughly equal in the latter years of observation. In the first, fourth and fifth years post-logging the differences in total vegetation cover were statistically significant. Herb cover was not consistently higher in either cable-yarded or tractor-logged plots and only the first year following logging was herb cover significantly higher in cable-yarded (15.4 percent) than tractor-logged plots (6.2 percent; two sample *t*-test, p-value <0.03).

Shrubs consistently had significantly higher cover in cable-yarded than tractor-logged plots throughout the post-logging period (Fig. 29). By contrast, overstory species had higher cover in tractor-logged than cable-yarded plots throughout the post-logging period (Fig. 30). In the seventh, eighth, tenth and thirty-fifth years post-logging the difference in overstory cover between tractor-logged and cable-yarded plots was statistically significant.

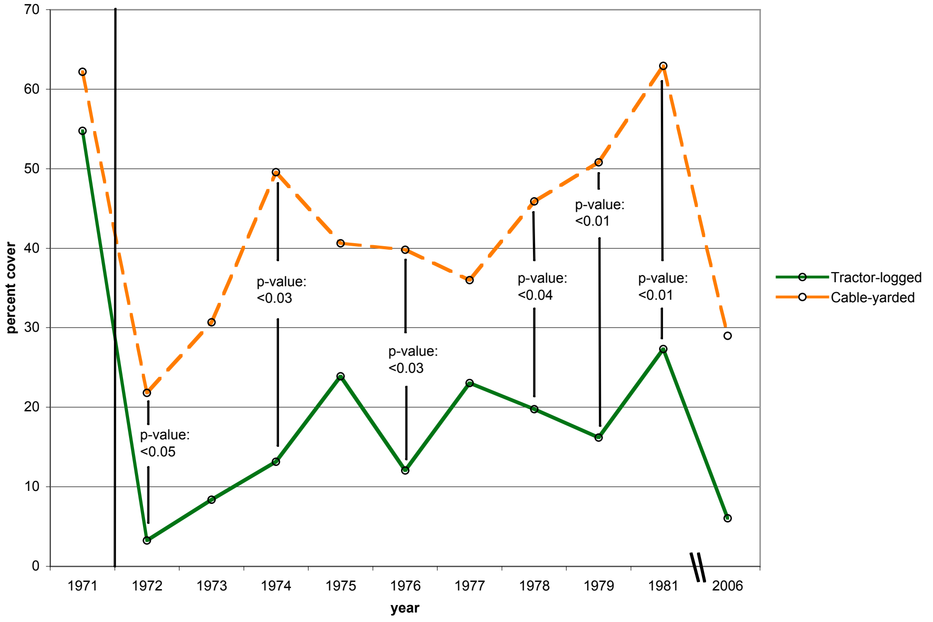

Fig. 29. Change in (tall + short) shrub cover in WS-2; plots stratified by logging method (n = 20 plots) (p-values from two sample *t*-test).

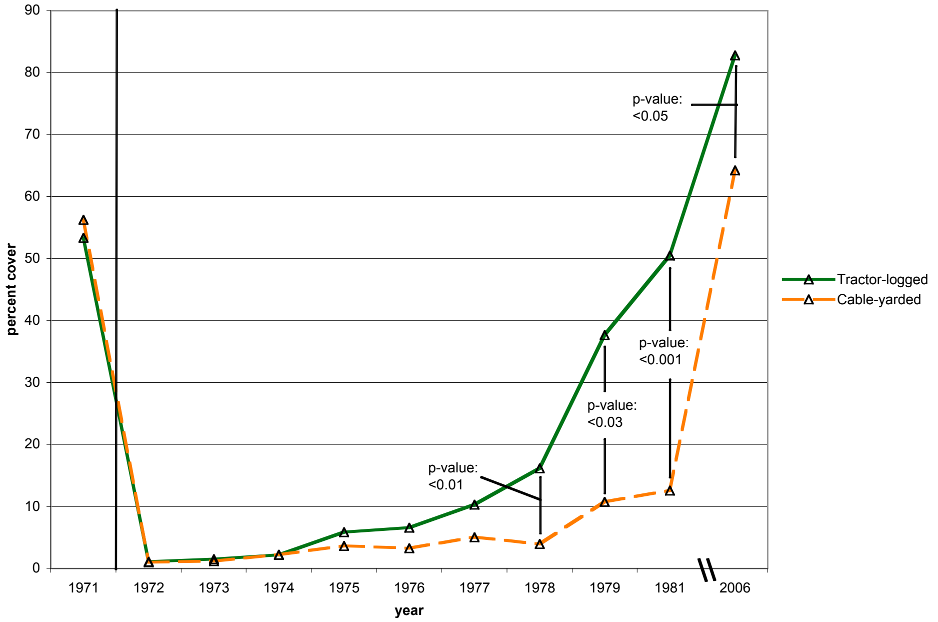

Fig. 30. Change in overstory canopy cover in WS-2; plots stratified by logging method (n = 20 plots) (p-values from two sample *t*-tests).

Relationship of soil disturbance to vegetation succession after patchcutting (WS-2) and clearcutting (WS-3)

Pre-logging data from plots stratified by post-logging soil disturbance showed total vegetation cover was similar on all plots of impending disturbance (99.6 to 100 percent) (Fig. 31). Following logging, total vegetation cover was highest on lightly disturbed plots, except in the tenth and thirty-fifth years when it was highest on moderately disturbed sites. In each year except the thirty-fifth following logging, total vegetation cover was significantly lower on heavily disturbed sites than either lightly or moderately disturbed plots. Total vegetation cover experienced significant declines on moderately and heavily disturbed plots in the fifth year post-logging. Thirty-five years post-logging, total vegetation cover was not significantly lower on heavily disturbed plots (92.2 percent) than either moderately or lightly disturbed plots (97.5 and 96.6 percent, respectively).

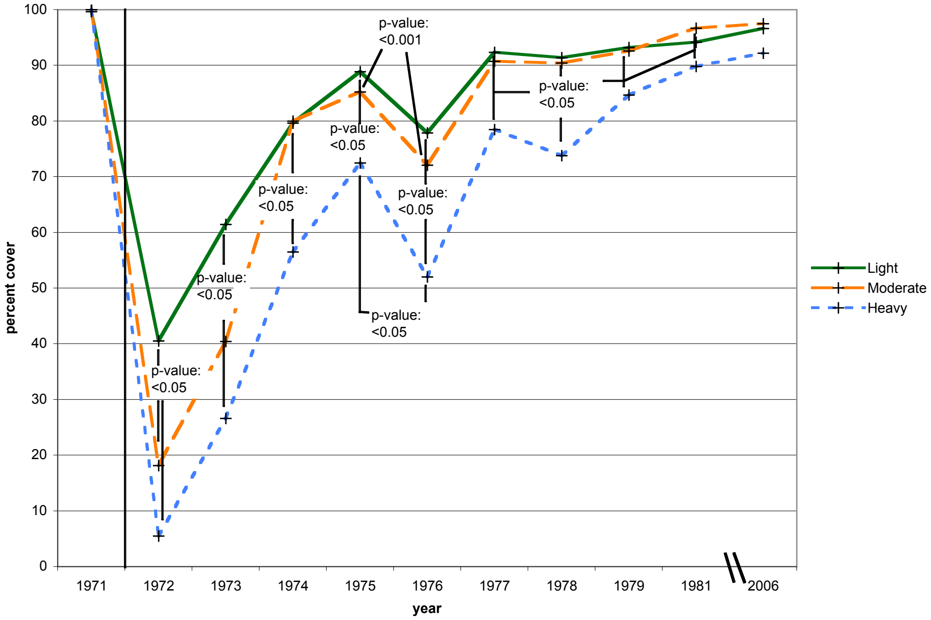

Fig. 31. Change in total vegetation cover in WS-2 & WS-3; plots stratified by soil disturbance (n = 36 plots) (difference p-values from ANOVA; decline p-values from two sample *t*-test).

In the pre-logging period, cover of overstory species was similar on all impending disturbance classes (50 to 54.2 percent). Cover of overstory species was increasingly higher on moderately and heavily disturbed sites than lightly disturbed plots; however, only the tenth year post-logging was the difference significant (ANOVA, p-value <0.05). In the thirty-fifth year post-logging, cover of overstory species was insignificantly lower (68.5 percent) than moderately and heavily disturbed plots (77.5 and 77 percent, respectively).

Prior to logging, shrub cover was insignificantly higher on plots that would become heavily disturbed (66.9 percent) than those plots that would become lightly or moderately disturbed (57.1 and 56.3 percent respectively) (Fig. 32). Following logging, shrub cover was highest on lightly disturbed sites and lowest on heavily disturbed plots except the thirty-fifth year when it was lowest on moderately disturbed plots. The difference was significant for each year except the third post-logging.

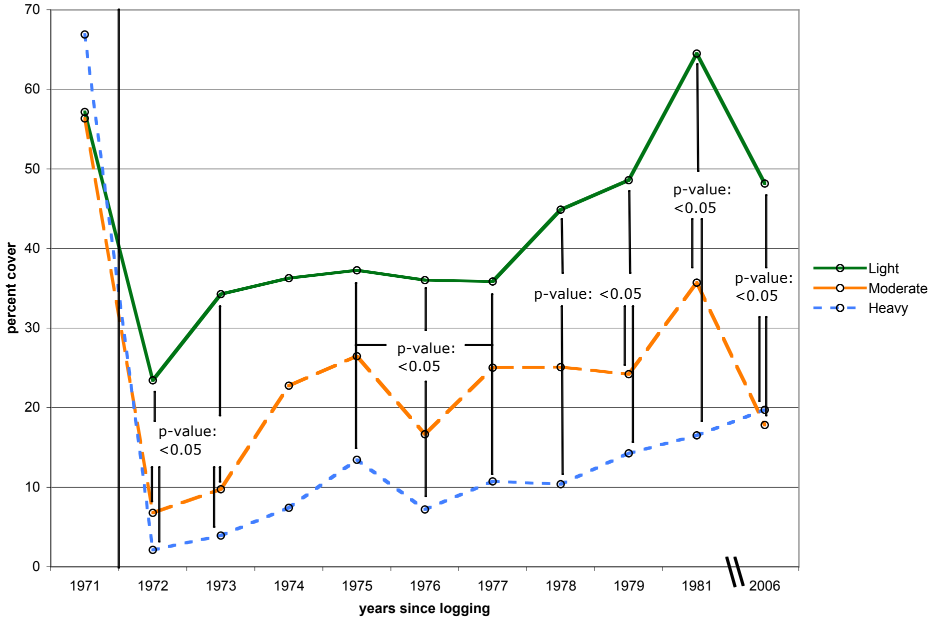

Fig. 32. Change in (tall + short) shrub cover in WS-2 & WS-3; plots stratified by soil disturbance (n = 36 plots) (p-values from ANOVA).

Prior to logging, herb cover was insignificantly higher on plots that would become heavily disturbed (32 percent) than on plots that would become moderately and lightly disturbed (29.1 and 27.6 percent, respectively). For the first six years following logging, herb cover was highest on lightly and moderately disturbed plots and highest on heavily disturbed plots the eighth and tenth years following logging. Only in the first and third years post-logging were differences in herb cover significant.

4.2 Change in Overstory Species

4.2.1 Effect of Thin, Clearcut and Small Patchcuts on Overstory Species Cover

In WS-2 and WS-3, of the seven overstory species, Douglas-fir had the highest cover (37.1 percent), followed by grand fir (8.5 percent), sugar pine (4 percent) and incense cedar (3.9 percent). Cover of western hemlock, Pacific madrone and ponderosa pine were each below one percent (0.8, 0.7 and 0.5 percent respectively).

Throughout the post-logging period, Douglas-fir had the highest cover (Fig. 33). Immediately following logging, grand fir and incense cedar had the second highest cover; however, in the sixth through eighth, tenth and thirty-fifth years, ponderosa pine had the second highest cover. Pacific madrone and western hemlock were not observed most years following logging. Sugar pine had the lowest cover throughout the post-logging period and was not observed in either WS-2 or WS-3 thirty-five years following logging.

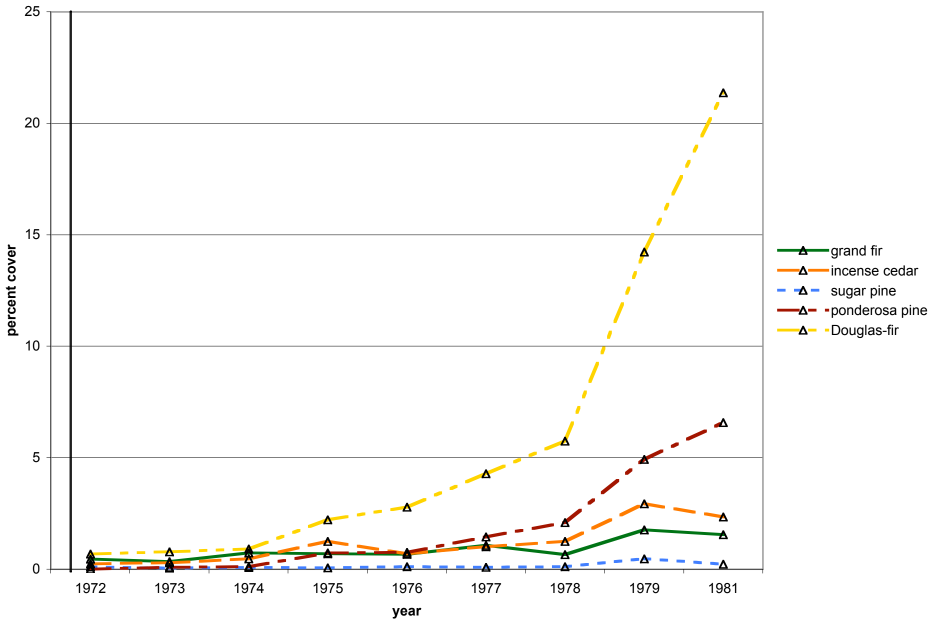

Fig. 33. Change in cover of 5 key overstory species in WS-2 + WS-3 (1971 & 2006 excluded for detail).

In WS-2 and WS-3, Douglas-fir, incense cedar and ponderosa pine significantly increased cover between 1971 and 2006 (Fig. 34). Grand fir, Pacific madrone, sugar pine and western hemlock declined in cover in the same period; however, only sugar pine's decline was statistically significant. When all three watersheds were averaged, again, Douglas-fir, incense cedar and ponderosa pine significantly increased cover in the same period. Western hemlock experienced an insignificant increase and grand fir achieved stasis. Only Pacific madrone and sugar pine declined; however, neither was statistically significant.

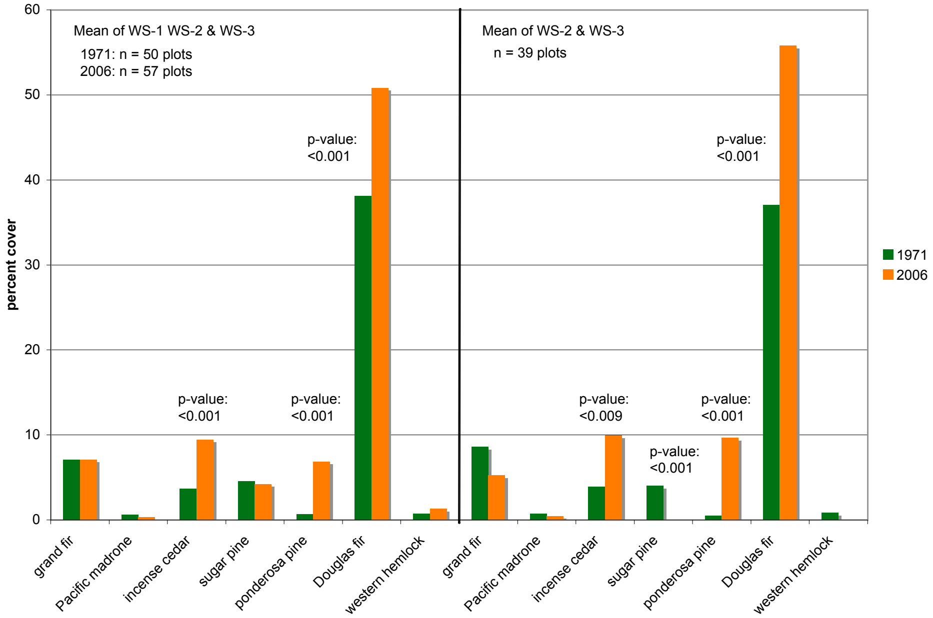

Fig. 34. Change in 7 overstory species in all three watersheds between 1971 & 2006 (p-values from two sample *t*-tests).

4.2.2 Relationship between Environmental Characteristics and Overstory Succession

Relationship of aspect to overstory species succession after patchcutting (WS-2) and clearcutting (WS-3)

Prior to logging, Douglas-fir, grand fir, incense cedar and western hemlock had the highest cover on north-facing sites. Douglas-fir and incense cedar had the lowest cover on northwest-facing sites, while grand fir had the lowest cover on east-facing sites. Pacific madrone and sugar pine had the highest cover on northeast-facing sites and lowest on north-facing sites. Ponderosa pine had the highest cover on northeast- and east-facing sites and no observations on the remaining sites.

Following logging, Douglas-fir, incense cedar and grand fir had the consistently highest cover on northwest-facing sites. Ponderosa pines had consistently highest cover on north-facing sites and the lowest cover on east- and southeast-facing sites. Sugar pine had consistently and periodically significantly higher cover on southeast sites than other aspects (Fig 35); however, sugar pine was not observed on any aspect thirty-five years after logging.

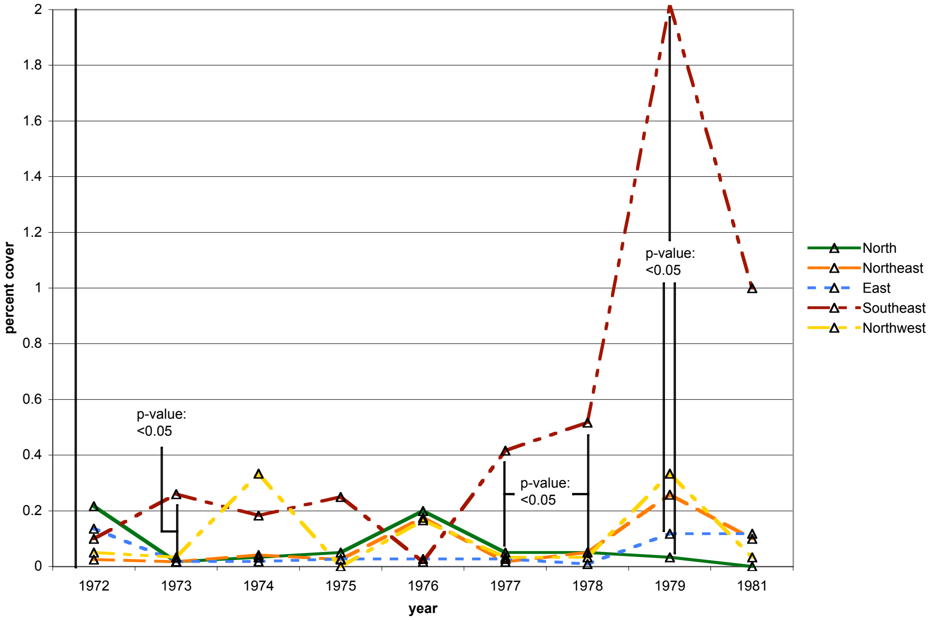

Fig. 35. Change in sugar pine cover in WS-2 & WS-3 (1971 excluded for detail); plots stratified by aspect (n = 39 plots) (p-values from ANOVA).

Relationship of slope to overstory species succession after patchcutting (WS-2) and clearcutting (WS-3)

Prior to logging, grand fir, Pacific madrone, incense cedar, sugar pine and western hemlock had the highest cover on steep slopes, while both Douglas-fir and ponderosa pine had the highest cover on moderate slopes. However, no species had significantly higher cover on any slope position.

Following logging, Douglas-fir had consistently higher cover on gradual slopes than either moderate or steep slopes with statistically significant differences between sites in the seventh, eighth and tenth years post-logging (Fig. 36). No other species had significantly higher cover on any slope position following logging. Ponderosa pine and incense cedar had higher covers on gradual slopes then either moderate or steep slopes, while sugar pine and grand fir did not have consistently higher cover on any slope position.

Thirty-five years following logging, grand fir, madrone and hemlock had higher cover on steep slopes than moderate or gradual slopes. Incense cedar and ponderosa pine had higher cover on gradual slopes than moderate or steep slopes. Douglas-fir had higher cover on moderate slopes than steep and gradual slopes.

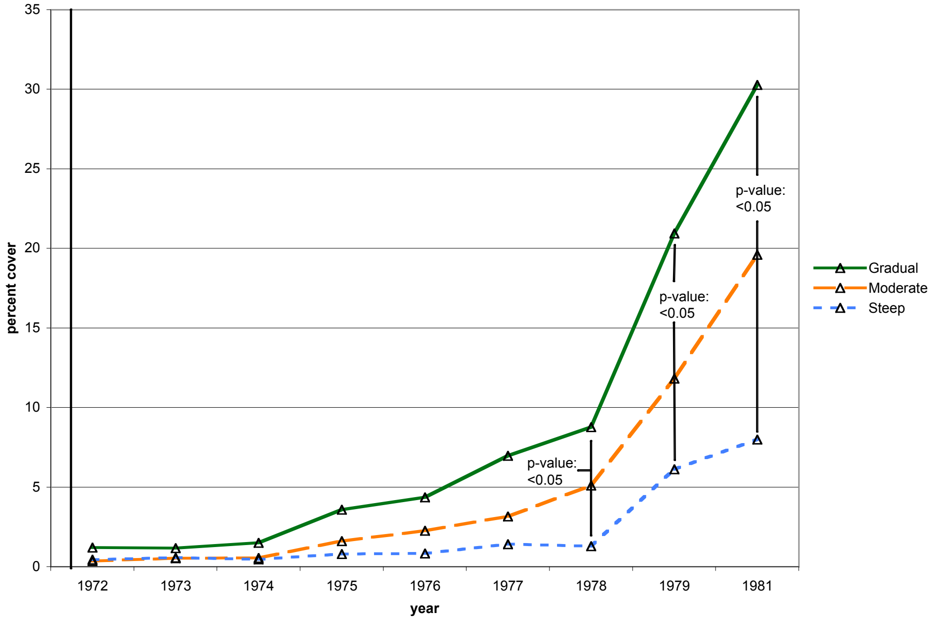

Fig. 36. Change in Douglas-fir cover in WS-2 & WS-3; plots stratified by slope (n = 39 plots) (p-values from ANOVA)

Relationship of soil series to overstory species succession after patchcutting (WS-2) and clearcutting (WS-3)

Prior to logging, grand fir, hemlock, ponderosa pine and madrone cover was higher on Dumont soils than Straight or other soils. Douglas-fir, sugar pine and incense cedar had higher cover on other soils than Dumont or Straight soils. All species except incense cedar had the lowest cover on Straight soils; however, no species had significantly higher cover on any soil series.

In the post-logging period, Douglas-fir, grand fir and sugar pine had consistently higher cover on Straight soils than Dumont or other soils. Ponderosa pine and incense cedar had consistently higher cover on both Dumont and Straight soils. During the period of observation, there were no statistically significant differences between soil types for any overstory species. Thirty-five years post-logging, ponderosa pine and incense cedar had the highest cover on Straight soils. Douglas-fir and grand fir had the highest cover on other soil types. Sugar pine and western hemlock were virtually absent from observation (0 and 0.6 percent respectively).

Relationship of vegetation community to overstory species succession after patchcutting (WS-2) and clearcutting (WS-3)

In the pre-logging community, the highest diversity of tree species (six of seven species) was in the Douglas-fir-grand fir/oceanspray-salal and unclassified communities; however, the highest total overstory cover was in the Douglas-fir/brome-whipplea community (Table 9). Conversely, the fewest number of tree species represented with three was in the Douglas-fir/brome-whipplea community, while the lowest total overstory cover was in the Douglas-fir-grand fir/oceanspray-salal community. Douglas-fir had the highest cover in all communities and was the only species observed in each community type, while ponderosa pine had the lowest cover and was present in only the Douglas-fir-grand fir/oceanspray-salal community.

With few exceptions, Douglas-fir continued to have the highest cover and frequency of the seven tree species in all community types. Five years post-logging grand fir had the highest cover in the Douglas-fir-grand fir/oceanspray-salal community, while five and ten years post-logging, ponderosa pine had the highest cover in the unclassified sites.

Table 9. Cover and frequency for 7 overstory species on 5 community types for pre-logging; five, ten and thirty-five years post-logging (red box denotes species highest cover/year; blue ellipse denotes species highest frequency/year; green star denotes community highest cover/year; blue star denotes subequal).

PBW

	1971		1976		1981		2006	
Overstory species	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**
grand fir	0	0	0.4	42.9	0.1	42.9	1	33.3
madrone	0	0	0	0	0	0	0	0
incense cedar	2.4	57.1	1.6	57.1	3.3	100	1.1	50
sugar pine	6.1	85.7	0.04	42.9	1	71.4	0	0
ponderosa pine	0	0	0.3	71.4	1.1	71.4	1.9	66.7
Douglas fir	52.8 *	100	6.2 *	100	29.5 *	100	78.3 *	100
western hemlock	0	0	0	0	0	0	0	0

PAHG

	1971		1976		1981		2006	
Overstory species	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**
grand fir	0.8	16.7	3.3 *	16.7	6.1	50	9.1	33.3
madrone	4.1	66.7	0.1	16.7	0	0	2.8	33.3
incense cedar	5.6	100	0.03	33.3	2	50	8.3	50
sugar pine	4	66.7	0.03	33.3	0.03	33.3	0	0
ponderosa pine	3.3	33.3	0.05	50	1.3	66.7	2.8	66.7
Douglas fir	27.5 *	100	2.8	83.3	12.3 *	83.3	45.3 *	100
western hemlock	0	0	0	0	0	0	0	0

PAG

	1971		1976		1981		2006	
Overstory species	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**
grand fir	2.5	37.5	0.1	75	1.2	87.5	9.5	37.5
madrone	0	0	0	0	0	0	0	0
incense cedar	1.2	25	0.4	75	1.4	75	8.7	75
sugar pine	6.8	62.5	0.3	12.5	0.2	25	0	0
ponderosa pine	0	0	0.5	25	4.5	75	3.2	62.5
Douglas fir	42.5 *	100	2.6 *	100	29.2 *	100	63.1 *	100
western hemlock	0	0	0	0	0	0	0	0

PTG

	1971		1976		1981		2006	
Overstory species	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**
grand fir	15.5	77.8 *	0.1	62.5	0.3	37.5	0.6	22.2
madrone	0	0	0	0	0	0	0	0
incense cedar	2.3	33.3	0.6	25	1.6	75	13.3	88.9 *
sugar pine	2.8	55.6	0.01	12.5	0	0	0	0
ponderosa pine	0	0	0.3	75	4.2	75	11.6	77.8
Douglas fir	28.3 *	100	1 *	100	22.5 *	100	59.4 *	100
western hemlock	2	33.3	0	0	0	0	0.1	11.1

Uncl.

	1971		1976		1981		2006	
Overstory species	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**	**cover**	**freq.**
grand fir	18.8	77.8 *	0.02	22.2	0.8	44.4	6.8	33.3
madrone	0.5	11.1	0	0	0	0	0.1	11.1
incense cedar	7.7	77.8	0.7	44.4	3.2	66.7	14.4	88.9 *
sugar pine	1.1	22.2	0.1	33.3	0.01	11.1	0	0
ponderosa pine	0	0	2.1 *	66.7	18.1 *	88.9	24	100
Douglas fir	35 *	100	1.7	88.9	13.1	100	37.7 *	100
western hemlock	1.6	11.1	0	0	0	0	0	0

Thirty-five years post-logging, Douglas-fir had the highest cover and within-community frequency. Incense cedar had the second highest cover, followed by ponderosa pine and grand fir. Grand fir, incense cedar, ponderosa pine and Douglas-fir were found in each community, while hemlock and madrone were found in only one and two types, respectively. Thirty-five years following logging, sugar pine was absent from all community types.

4.2.3 Relationship between Disturbance and Overstory Succession

Relationship of logging treatment to overstory species succession after patchcutting (WS-2) and clearcutting (WS-3)

Although there were no significant differences in the overstory cover between WS-2 and WS-3, there were significant differences among individual species between the watersheds. Douglas-fir (41.9 and 31.4 percent) and sugar pine (5.8 and 1.9 percent) had significantly higher cover in WS-2 than WS-3 prior to logging (two sample *t*-test, p-value <0.04 and <0.03 respectively). Conversely, western hemlock (1.8 and 0 percent) and grand fir (13.1 and 4.7 percent) had higher cover in WS-3 than WS-2, though these differences were not significant. Incense cedar (4.5 and 3.3 percent) and madrone (1.1 and 0.4 percent) had insignificantly higher cover in WS-3 than WS-2, while the inverse was true for ponderosa pine (0 and 0.9 percent).

Thirty-five years after logging, the cover of grand fir, western hemlock and madrone was roughly equivalent in both watersheds. Douglas-fir had higher cover in WS-2 than WS-3 throughout the later post-logging years, with the thirty-fifth year significantly different (62.7 and 47.8 percent, respectively; two sample *t*-test, p-value <0.03). Incense cedar had insignificantly higher cover in WS-2 than WS-3 after logging, except the thirty-fifth year when the cover was significantly higher in WS-3 than WS-2 (two sample *t*-test, p-value <0.03). Ponderosa pine had increasingly higher cover in WS-3 than WS-2 with the eighth, tenth and thirty-fifth years significantly different (Fig. 37).

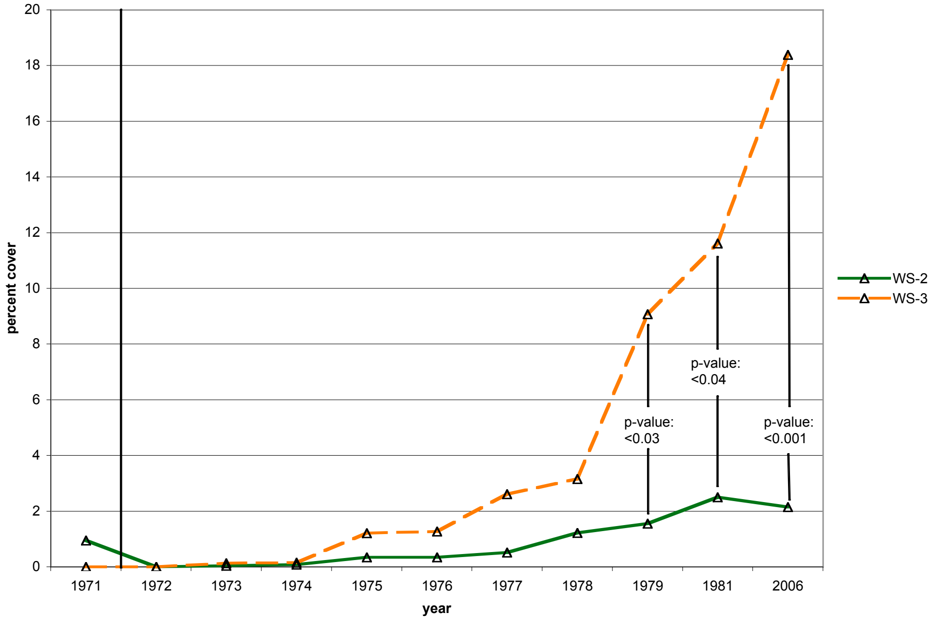

Fig. 37. Change in ponderosa pine cover in WS-2 & WS-3; plots stratified by logging treatment (n = 39 plots) (p-values from two sample *t*-tests).

Considering all three treated watersheds in 1971 and 2006, grand fir cover significantly increased in WS-1 and declined in WS-3. Grand fir cover increased in WS-2, but was not significant (Table 10). Pacific madrone cover showed an insignificant increase in WS-1 and insignificant declines in WS-2 and WS-3. Incense cedar increased in cover in all three watersheds, with only increases in WS-1 and WS-3 being significant.

Sugar pine declined significantly in WS-2 and WS-3 and insignificantly in WS-1. Neither the increase in ponderosa pine cover in WS-2 nor the decrease in WS-1 was significant; however, the increase in ponderosa pine cover in WS-3 was significant.

Douglas-fir cover showed an insignificant decline in WS-1 and significant increases in both WS-2 and WS-3 from 1971 to 2006. Western hemlock cover had insignificant increases in both WS-1 and WS-2, but experienced a significant decline in WS-3.

Table 10. Change in seven overstory canopy species percent cover in WS-1; WS-2; WS-3 and between watersheds, 1971 & 2006 (1971: n = 50 plots; 2006: n = 57 plots) (p-values from two sample *t*-tests).

		Grand fir	Pacific madrone	Incense cedar	Sugar pine	Ponderosa pine	Douglas-fir	Western hemlock	Total
WS-1	1971	1.8	0.0	2.9	6.4	1.4	41.8	0.3	49.5
	2006	11.1	0.02	8.4	13.3	0.8	40.0	4.2	66.1
	change	9.3	0.02	5.5	6.9	-0.5	-1.8	3.9	16.6
	p-value	0.0115	0.6126	0.0145	0.2276	0.6630	0.7454	0.0928	0.0122
WS-2	1971	5.0	0.5	3.0	6.1	1.0	40.5	0.0	53.8
	2006	5.5	0.2	5.6	0.0	2.3	63.1	0.03	72.8
	change	0.5	-0.4	2.6	-6.1	1.3	22.6	0.03	19.1
	p-value	0.9129	0.2725	0.3287	0.0001	0.1892	0.0007	0.3236	0.0016
WS-3	1971	13.1	1.1	4.6	2.0	0.0	31.4	1.8	48.6
	2006	5.3	0.8	14.7	0.0	18.4	47.8	0.0	76.7
	change	-7.8	-0.3	10.2	-2.0	18.4	16.4	-1.8	28.1
	p-value	0.0648	0.8183	0.0071	0.0216	0.0000	0.0033	0.0000	0.0000
Canopy	1971	-3.2	-0.5	-0.1	0.3	0.4	1.3	0.3	-1.5
Diff. b/t	p value	0.526	0.230	0.958	0.915	0.736	0.817	0.182	0.817
WS-1 &	2006	5.6	-0.1	2.7	13.3	-1.4	-23.1	4.1	1.2
WS-2	p value	0.147	0.309	0.351	0.001	0.179	0.001	0.016	0.856
	change	8.8	0.4	2.8	13.0	-1.8	-24.4	3.9	2.7
Canopy	1971	-11.2	-1.1	-1.7	4.4	1.4	10.4	-1.6	0.7
Diff. b/t	p value	0.006	0.330	0.431	0.087	0.019	0.078	0.237	0.929
WS-1 &	2006	5.9	-0.8	-6.4	13.3	-17.6	-7.8	4.2	-9.1
WS-3	p value	0.144	0.341	0.082	0.002	0.000	0.135	0.021	0.181
	change	17.1	0.3	-4.7	8.9	-18.9	-18.2	5.7	-9.8
Canopy	1971	-8.1	-0.6	-1.6	4.1	1.0	9.1	-1.8	2.2
Diff. b/t	p value	0.088	0.489	0.361	0.016	0.177	0.065	0.058	0.702
WS-2 &	2006	0.2	-0.7	-9.1	0.0	-16.1	15.3	0.0	-10.4
WS-3	p value	0.957	0.397	0.030	1.000	0.000	0.026	0.350	0.155
	change	8.3	-0.1	-7.5	-4.1	-17.1	6.2	1.9	-12.5

Considering all three treated watersheds, grand fir sapling cover declined significantly in WS-2 and WS-3, but declined insignificantly in WS-1 (Table 11). The largest difference in grand fir sapling cover change occurred between WS-2 and WS-3 (9.8 percent). Pacific madrone sapling cover increased in WS-3 only; however, this change was not significant. Cover of incense cedar saplings insignificantly declined in both WS-1 and WS-2 and insignificantly increased in WS-3. The largest difference in incense cedar cover change occurred between WS-1 and WS-3 (5.3 percent).

Sugar pine sapling cover increased in WS-1, but declined in significantly in WS-2 and insignificantly in WS-3. Ponderosa pine sapling cover increased insignificantly in all three watersheds. Cover of Douglas-fir increased significantly in WS-1, but declined insignificantly in WS-2 and WS-3. The largest difference in Douglas-fir sapling cover occurred between WS-1 and WS-3 (21.8 percent) followed closely by the difference between WS-1 and WS-2 (20.1 percent). Western hemlock sapling cover had an insignificant increase in cover in WS-1 and decline in WS-2, but experienced a significant increase in WS-3. The difference largest difference between in western hemlock sapling cover occurred between WS-1 and WS-3 (19.1 percent).

Table 11. Change in seven overstory sapling species percent cover in WS-1; WS-2; WS-3 and between watersheds, 1971 & 2006 (1971: n = 50 plots; 2006: n = 57 plots) (p-values from two sample *t*-tests).

		Grand fir	Pacific madrone	Incense cedar	Sugar pine	Ponderosa pine	Douglas-fir	Western Hemlock	Total
WS-1	1971	17.7	0	9.2	0.3	0.2	3.6	0.9	31.9
	2006	7.8	0	4.8	0.5	0.6	22.5	4.2	40.5
	change	-9.9	0	-4.3	0.2	0.4	18.9	3.2	8.6
	p-value	0.1256	0.0	0.1625	0.7067	0.5597	0.0031	0.3210	0.4016
WS-2	1971	11.0	0	5.8	0.2	0.0	4.7	1.1	22.8
	2006	1.3	0	2.3	0.0	0.3	3.5	0.0	7.4
	change	-9.8	0	-3.5	-0.2	0.3	-1.2	-1.1	-15.4
	p-value	0.0019	0.0	0.1945	0.0187	0.1969	0.6160	0.1730	0.0025
WS-3	1971	20.2	0.1	1.1	0.3	0.0	4.8	15.9	42.4
	2006	0.6	0.2	2.1	0.0	0.3	1.9	0.0	5.1
	change	-19.6	0.1	1.0	-0.3	0.3	-2.9	-15.9	-37.3
	p-value	0.0007	0.7832	0.2192	0.0845	0.3145	0.4366	0.0103	0.0000
Sapling Diff. b/t WS-1 & WS-2	1971	6.7	0	3.4	0.1	0.2	-1.1	-0.2	9.1
	p value	0.2853	0.0	0.4102	0.5467	0.2074	0.6641	0.8820	0.1943
	2006	6.6	0	2.6	0.5	0.3	19.1	4.1	33.1
	p value	0.0262	0.0	0.1211	0.1574	0.6473	0.0002	0.0808	0.0000
	change	-0.1	0	-0.9	0.4	0.1	20.1	4.3	24.0
Sapling Diff. b/t WS-1 & WS-3	1971	-2.4	-0.1	8.0	0.0	0.2	-1.2	-15.0	-10.5
	p value	0.7765	0.4444	0.0063	0.8902	0.2064	0.8034	0.0584	0.3014
	2006	7.2	-0.2	2.7	0.5	0.3	20.6	4.2	35.4
	p value	0.0145	0.3244	0.0440	0.1763	0.6000	0.0001	0.0960	0.0000
	change	9.6	-0.1	-5.3	0.5	0.1	21.8	19.1	45.9
Sapling Diff. b/t WS-2 & WS-3	1971	-9.1	-0.1	4.6	-0.1	0.0	-0.1	-14.8	-19.6
	p value	0.1216	0.2981	0.0862	0.6886	0.1772	0.9723	0.0122	0.0122
	2006	0.6	-0.2	0.2	0.0	0.1	1.6	0.0	2.3
	p value	0.5644	0.2981	0.8917	0.3498	0.8802	0.4176	0.3498	0.4964
	change	9.8	-0.1	-4.5	0.1	0.0	1.7	14.8	21.9

Changes in WS-4 between 1971 and 2006 are unknown because data was not collected in 1971. In 2006, all seven species were observed in WS-4 (Fig. 38). Douglas-fir cover was the highest of all species (41.5 percent), while grand fir (17.3 percent), incense cedar (11.5 percent) and western hemlock (5.3 percent) cover were the consecutively next highest. Ponderosa pine, sugar pine and madrone cover were the lowest (1.3, 0.5 and 0.3 percent, respectively).

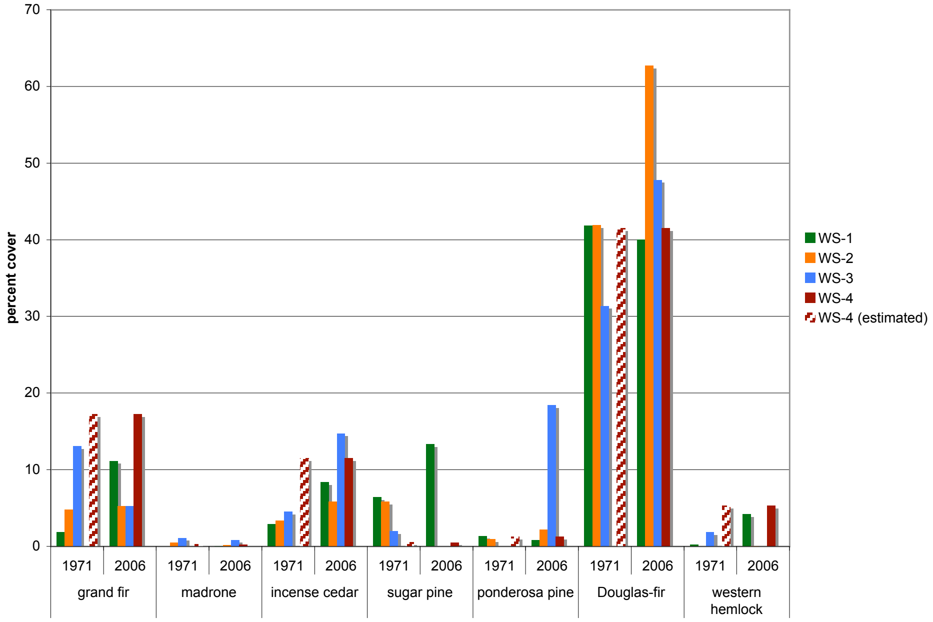

Fig. 38. Total cover in 1971 & 2006 of overstory canopy species of WS 1-4; plots stratified by logging treatment (1971: n = 50 plots; 2006: n = 57 plots (n = 77 plots including WS-4)); data for WS-4 in 1971 estimated from 2006 surveys

Relationship of in-watershed location to overstory species succession after clearcutting (WS-3)

In the pre-logging period, western hemlock, sugar pine and madrone had higher cover in center plots, while ponderosa pine, Douglas-fir and incense cedar had higher cover in edge plots; however no differences were statistically significant. Following logging, all overstory species had higher cover on edge sites than center sites. Only ponderosa pine and incense cedar were statistically significant in the thirty-fifth year (two sample *t*-test, p-value <0.05 and <0.02 respectively). Western hemlock and madrone were not observed until the thirty-fifth year.

Relationship of logging method to overstory species succession after patchcutting (WS-2)

Prior to logging, no overstory species cover was significantly different between plots that would be tractor-logged compared to plots that would be cable-yarded. Following logging, grand fir and sugar pine had insignificantly higher cover on cable-yarded plots than tractor-logged plots; however, sugar pine was no longer observed thirty-five years after logging. In the post-logging period, incense cedar, ponderosa pine and Douglas-fir had higher cover on tractor-logged plots; however, only ponderosa pine and Douglas-fir were statistically significant (Figs. 39 and 40).

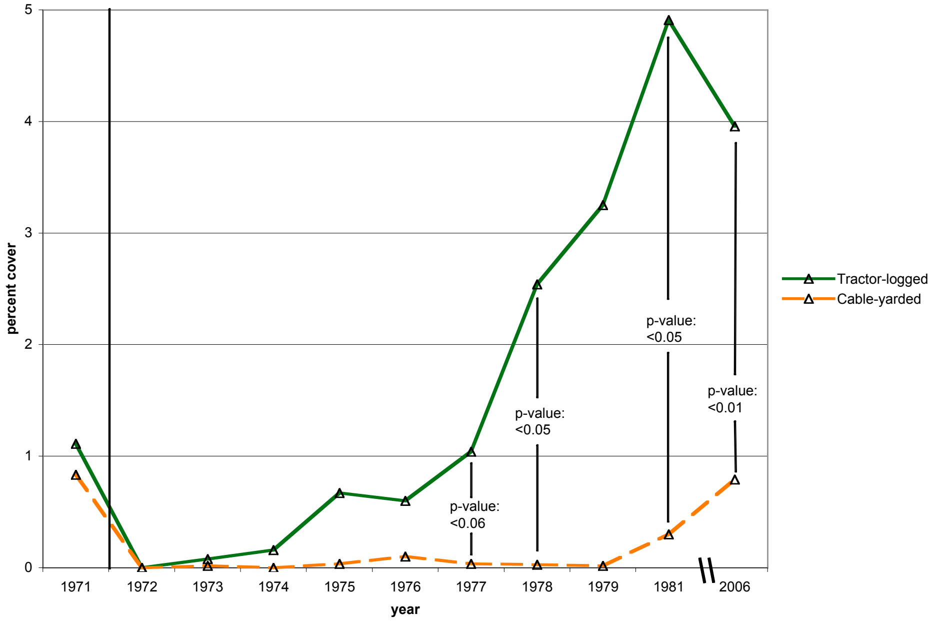

Fig. 39. Change in ponderosa pine cover in WS-2; plots stratified by logging method (n = 20 plots) (p-values from two sample *t*-test).

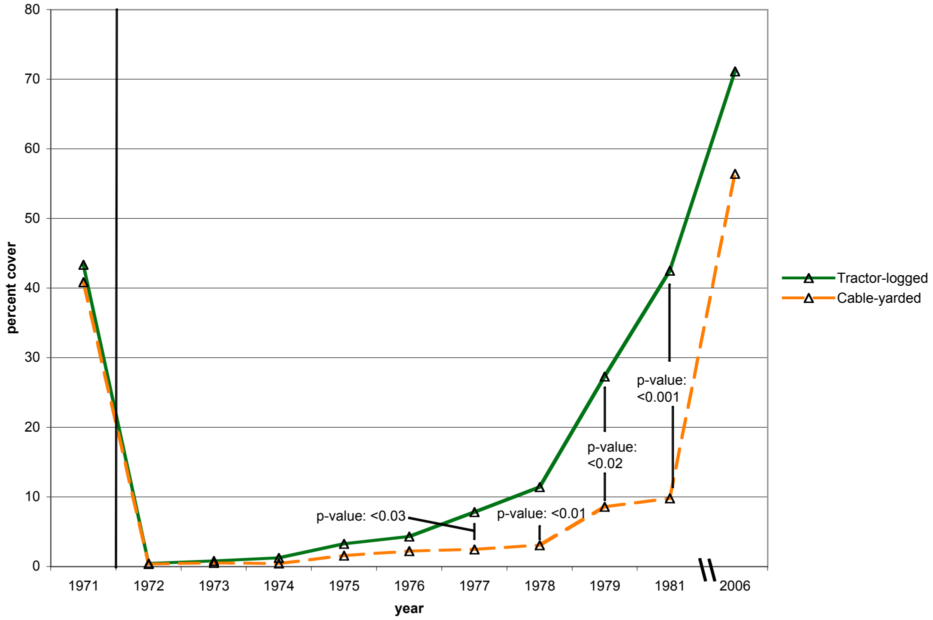

Fig. 40. Change in Douglas-fir cover in WS-2; plots stratified by logging method (n = 20 plots) (p-values from two sample *t*-tests).

Relationship of soil disturbance to overstory species succession after patchcutting (WS-2) and clearcutting (WS-3)

Pre-logging data from plots stratified by post-logging soil disturbance showed overstory tree species cover was similar on all sites of impending disturbance. During the post-logging period, incense cedar consistently had the highest cover on moderately disturbed plots and the lowest cover on heavily disturbed plots. Grand fir consistently had the highest post-logging cover on lightly disturbed plots, and similar cover on heavily and moderately disturbed plots. Sugar pine had roughly equivalent cover throughout the post-logging period, with no observations in the thirty-fifth year on any of the three classes of soil disturbance. Western hemlock and madrone were not observed until the thirty-fifth year, with hemlock found on lightly disturbed plots only and madrone found on plots in all three disturbance classes.

Only ponderosa pine and Douglas-fir had statistically significant differences between the soil disturbance sites in the post-logging period. Both species had higher cover on heavily and moderately disturbed sites than lightly disturbed sites with significant differences in later years (Figs. 41 and 42).

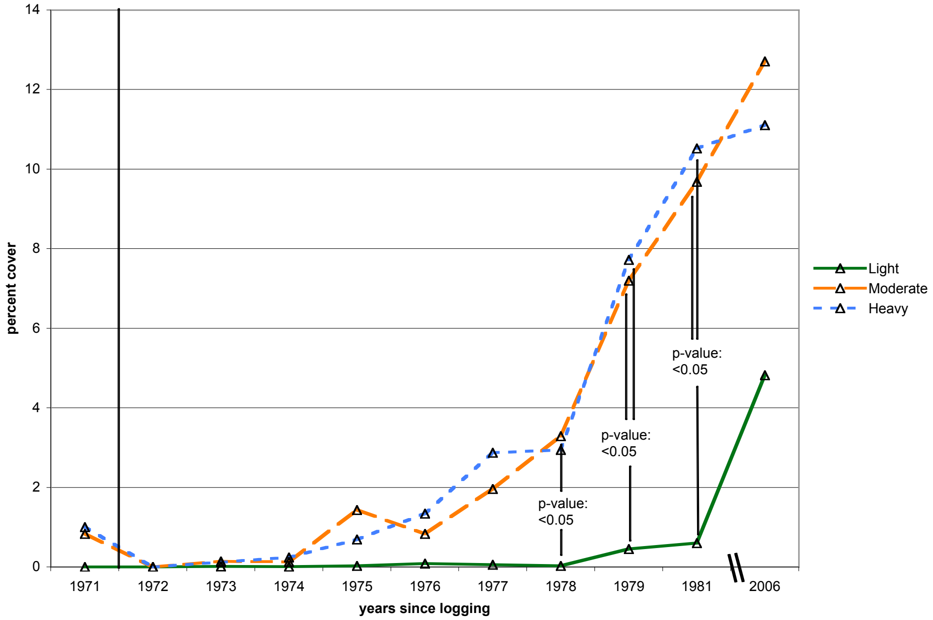

Fig. 41. Change in ponderosa pine cover in WS-2 & WS-3; plots stratified by soil disturbance (n = 36 plots) (p-values from ANOVA)

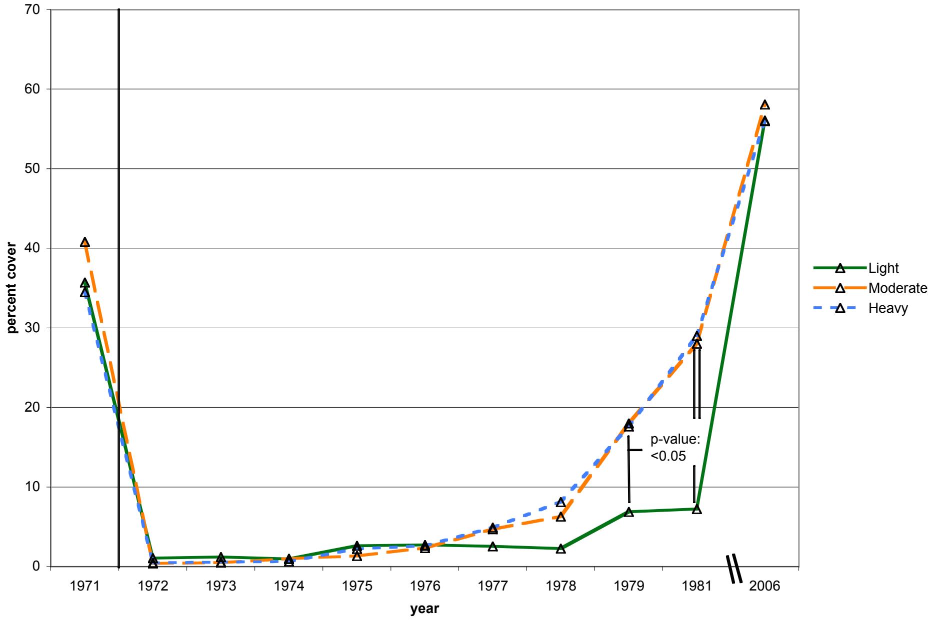

Fig. 42. Change in Douglas-fir cover in WS-2 & WS-3; plots stratified by soil disturbance (n = 36 plots) (p-values from ANOVA).

4.3 <u>Change in Residual and Invader Species</u>

4.3.1 Effect of Clearcut and Small Patchcuts on Residual and Invader Species Cover

In the pre-logging period, both residual shrubs and herbs were significantly higher in WS-2 and WS-3 than invader shrubs and herbs (Fig. 43). Throughout the post-logging period, residual shrubs had significantly higher cover than invader shrubs. Conversely, invader herbs had significantly higher cover than residual herbs except in the first, sixth and thirty-fifth years post-logging. Only net values of residual and invader shrubs and herbs will be presented.

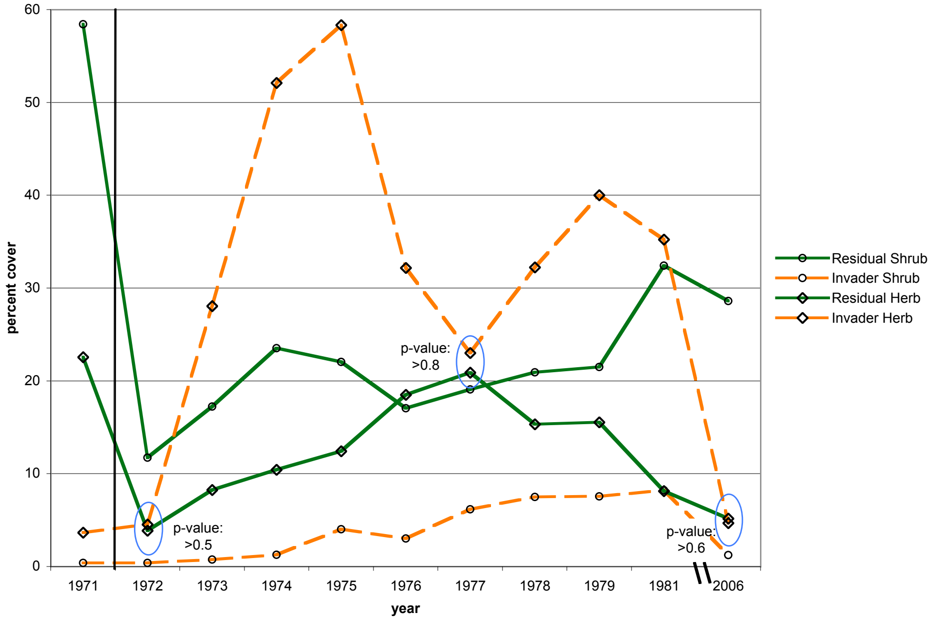

Fig. 43. Change in residual shrubs & herbs and invader shrubs & herbs in WS-2 + WS-3 (n = 39 plots). Blue ellipse denotes statistically insignificant differences (p-values from two sample *t*-tests).

After the initial decline in cover sustained from logging, residual shrubs increased or maintained cover throughout the period of observation, declining somewhat in the thirty-fifth year (Fig. 43). *Rubus ursinus* was the only species that increased cover immediately following logging; however, its cover had decreased by the thirty-fifth year, along with *Rosa gymnocarpa* (Fig. 44). *Gaultheria shallon*, *Berberis nervosa* and *Acer circinatum* maintained a low level of cover throughout the first ten years of observation, but by the thirty-fifth year had increased cover.

Invader shrubs increased cover following logging, peaking in the tenth year (Fig. 43). The key invader shrub species gained cover at roughly the same periods and by the thirty-fifth year post-logging had all decreased (Fig. 45). Both *Rubus leucodermis* and *Ceanothus* spp cover (*C. integerrimus*, *C. sanguineus* and *C. velutinus*) declined to approximate pre-logging levels by the thirty-fifth year post-logging, while *Symphoricarpos mollis* cover was still higher than its pre-logging level.

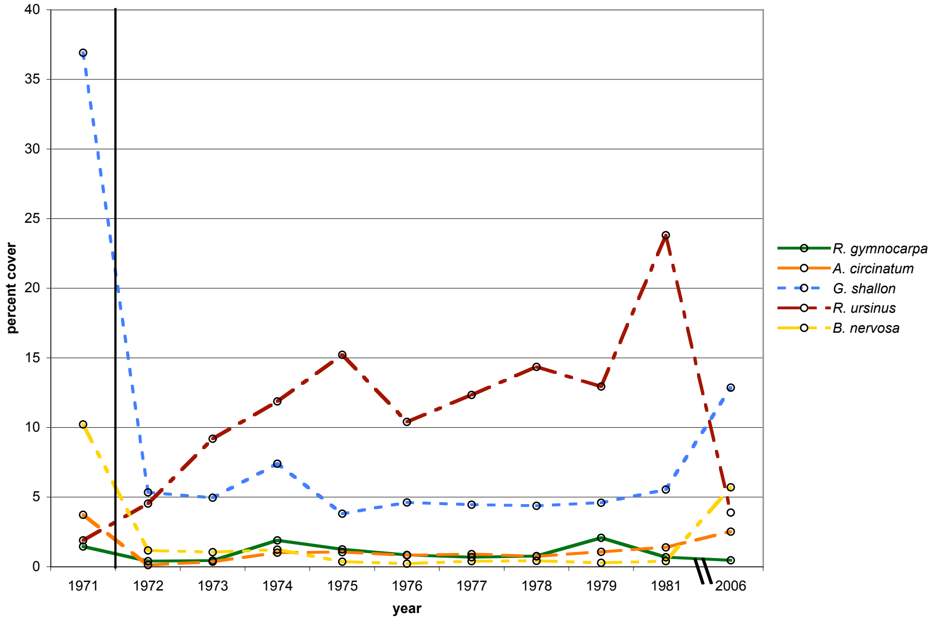

Fig. 44. Change in five key residual shrub species in WS-2 + WS-3 (n = 39 plots).

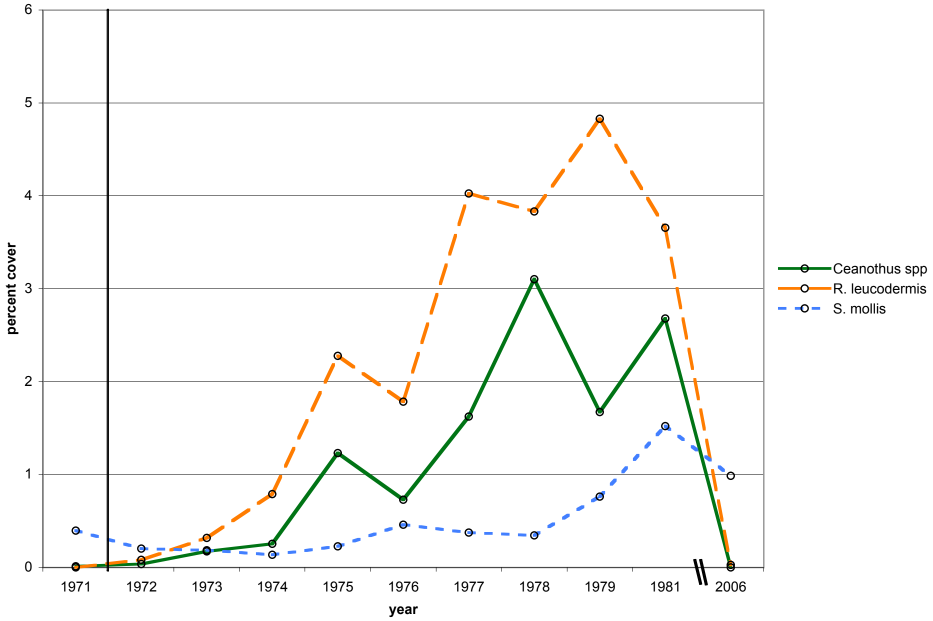

Fig. 45. Change in five key invader shrubs species in WS-2 + WS-3 (n = 39 plots).

Invader herbs experienced two peaks following logging (Fig. 43); however, individual invader species peaked at different times (Figs. 46 and 47). The most remarkable was *Senecio sylvaticus*, which peaked the second year after logging then declined to no recorded observations by the seventh year. *Lactuca serriola* cover peaked in the third year. *Epilobium paniculatum* and *Madia madioides* cover each peaked twice: in the fourth and fifth years and again in the eighth and seventh years, respectively. *Agoseris aurantica* and *Cirsium vulgare* cover peaked once, each in the fourth year. The cover of the legumes, *Vicia americana* and *Lathyrus polyphyllus* and the fern, *Pteridium aquilinum*, peaked in the eighth and tenth years respectively, following the second peaks of *M. madioides* and *E. paniculatum*. As a group, invader herbs had declined to approximate pre-logging levels by the thirty-fifth year post-logging (3.6 to 4.6 percent, respectively); however some species, such as *P. aquilinum* maintained higher levels of cover thirty-five years after logging than in the pre-logging period.

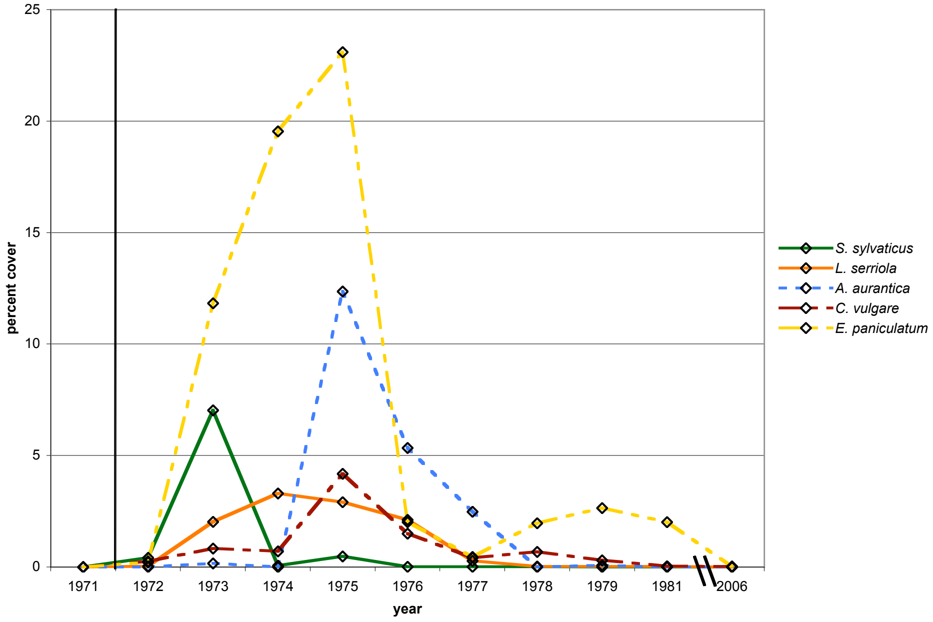

Fig. 46. Change in five key "early" invader herb species in WS-2 + WS-3 (n = 39 plots).

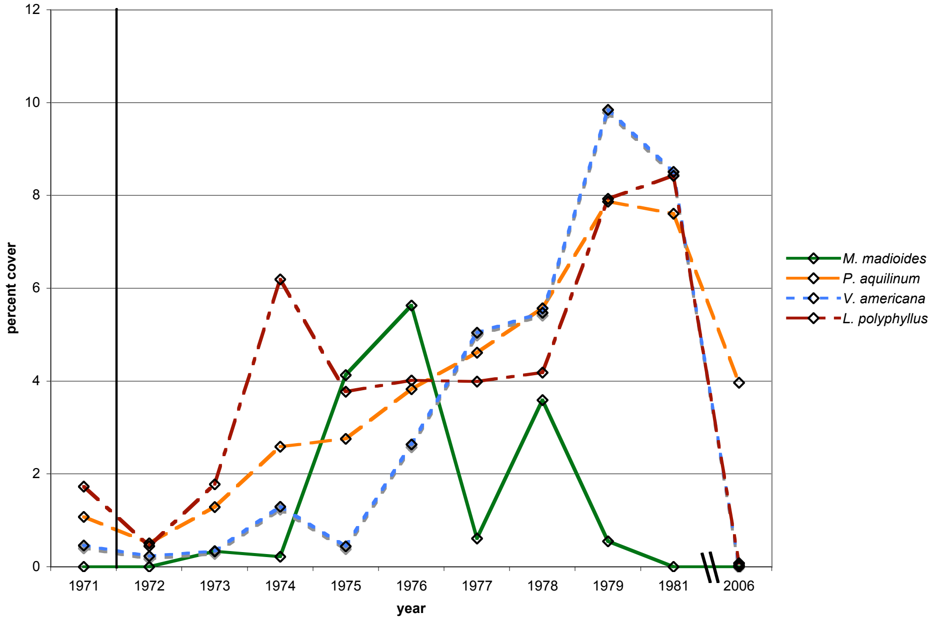

Fig. 47. Change in four key "late" invader herb species in WS-2 + WS-3 (n = 39 plots).

Residual herbs showed only one peak in cover (Fig. 43) and, as with invader herbs, this peak was not composed of all species equally (Fig. 48). Cover of most residual herbs declined after logging; however, *Whipplea modesta* cover was highest in the sixth year. All residual herbs declined to covers below their pre-logging levels.

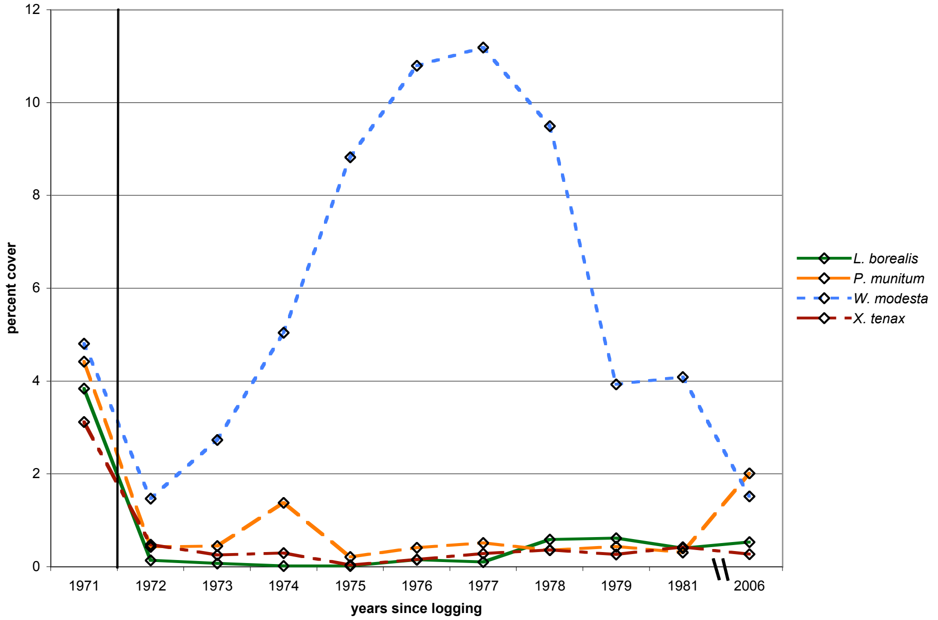

Fig. 48. Change in four key residual herb species in WS-2 + WS-3 (n = 39 plots).

4.3.2 Relationship between Environmental Characteristics and Residual and Invader Succession

Relationship of aspect to residual and invader species after patchcutting (WS-2) and clearcutting (WS-3)

Prior to logging, residual shrubs cover was highest on east-facing sites (67.3 percent) and lowest on southeast-facing sites (25.4 percent), while residual herbs cover was highest on southeast-facing sites (42.9 percent) and lowest on east-facing sites (20 percent). Neither set of residuals exhibited significant differences. Both invader shrubs and invader herbs had similar populations on each aspect without significant differences.

After logging, residual shrubs consistently had the lowest cover on southeast-facing and northeast-facing sites and the highest cover on east- and northwest-facing sites, while residual herbs had the lowest cover on northeast- and north-facing sites and the highest cover on southeast- and east-facing sites. Both invader shrubs and herbs had the lowest cover on northwest-facing sites and the highest cover on southeast- and east-facing sites. Periodically, the total invader population (shrubs + herbs) was significantly higher on southeast-facing sites than northwest-facing sites (Fig. 49).

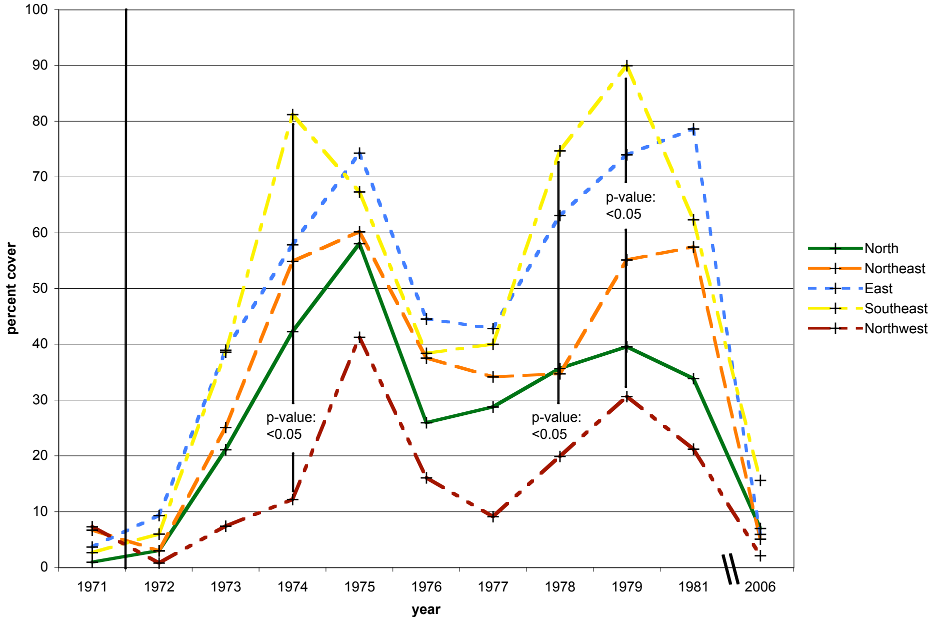

Fig. 49. Change in cover of all invader species (shrubs + herbs) in WS-2 & WS-3; plots stratified by aspect (n = 39 plots) (p-values from ANOVA).

Relationship of slope to residual and invader species succession after patchcutting (WS-2) and clearcutting (WS-3)

In the pre-logging period, residual shrubs and herbs had the highest cover on gradual slopes (63.1 and 36.5 percent respectively) and the lowest cover on moderate and steep slopes, respectively; however, these differences were not significant. Invader shrubs had similar cover on all slopes (0 to 0.5 percent), while invader herbs had significantly higher cover on gradual slopes (5.6 percent) than steep slopes (0.2 percent).

Most years following logging, both residual shrubs and herbs had higher cover on steep slopes. Only in the second and thirty-fifth years after logging were these differences significant for herb and shrub cover, respectively (ANOVA, p-value <0.05). Invader shrubs consistently had the highest cover on moderate slopes; however at no point were the differences significant. Invader herbs consistently had the highest cover on moderate slopes and the lowest on steep slopes (Fig. 50).

Fig. 50. Change in invader herb cover in WS-2 & WS-3; plots stratified by slope (n = 39 plots) (p-values from ANOVA).

<u>Relationship of soil series to residual and invader species after patchcutting (WS-2) and clearcutting (WS-3)</u>

Prior to logging, residual shrubs had significantly higher cover on Dumont soils (72.5 percent) than other soils (13.6 percent; ANOVA, p-value <0.05). Residual herbs had insignificantly higher cover on Dumont soils (23.7 percent) than either Straight or other soils (20.9 and 21.7 percent, respectively). Invader shrubs had the highest cover on other soils (20.9 and 21.7 percent, respectively) and the lowest cover on Dumont and Straight soils (0.3 percent each). Invader herbs had the highest cover on Straight soils (4.1 percent) and the lowest on other soils (3.2 percent). Neither invader shrub nor herb cover differences were significant.

Following logging, residual shrub or herb cover was neither consistently nor significantly higher on any soil type. Most years following logging, invader shrubs and herbs had slightly higher cover on Straight soils and on Dumont or other soils; however, at no point were the differences significant.

Relationship of vegetation community to residual and invader species after patchcutting (WS-2) and clearcutting (WS-3)

Prior to logging, residual shrubs exhibited statistically higher cover in Douglas-fir-grand fir/salal, Douglas-fir-grand fir/oceanspray-salal and Douglas-fir-western hemlock/salal communities (87.3, 86.5 and 82.3 percent, respectively) than Douglas-fir/brome-whipplea and unclassified communities (7.8 and 29.5 percent, respectively; ANOVA, p-value <0.05). Residual herbs had insignificantly higher cover in Douglas-fir-western hemlock/salal and Douglas-fir/brome-whipplea communities (27.5 and 26.8 percent respectively) than in Douglas-fir-grand fir/oceanspray-salal and Douglas-fir-grand fir/salal communities (18.6 and 17.3 percent, respectively). Invader shrubs had roughly equivalent cover in all community types (0.1 to 0.8 percent), while invader herbs had insignificantly higher cover in Douglas-fir-grand fir/oceanspray-salal and Douglas-fir/brome-whipplea communities (6.9 and 6.5 percent, respectively) than in the Douglas-fir-western hemlock/salal community (0.1 percent).

Following logging, residual shrubs were highest in the Douglas-fir-grand fir/oceanspray-salal community and lowest in Douglas-fir/brome-whipplea and unclassified communities. Residual herbs were highest in Douglas-fir/brome-whipplea and Douglas-fir-grand fir/salal communities and lowest in unclassified sites. Only in the sixth year following logging was either of these differences significant (ANOVA, p-value: <0.05). Invader shrubs consistently had the highest cover in Douglas-fir-grand fir/salal community and the lowest cover in Douglas-fir-western hemlock/salal and Douglas-fir-grand fir/oceanspray-salal communities; however the only significant difference was in the sixth year (ANOVA, p-value <0.05). Invader herbs had the consistently lowest cover in the Doulgas fir-western hemlock/salal community, but cover was consistently highest in no one community.

4.3.3 Relationship between Disturbance and Residual and Invader Succession

Relationship of logging treatment to residual and invader species after patchcutting (WS-2) and clearcutting (WS-3)

Residual shrubs had similar cover in each watershed prior to logging, while residual herbs had insignificantly higher cover in WS-2 (28.5 percent) than WS-3 (27.5 percent). Both invader shrubs and herbs had insignificantly higher cover in WS-2 (0.7 and 5.5 percent, respectively) than WS-3 (0.1 and 2 percent, respectively).

Each year following logging, residual shrubs had higher cover in WS-2 than WS-3 except the thirty-fifth year. At no point was the difference between WS-2 and WS-3 statistically significant except the third year (36.4 and 10.8 percent, respectively; two sample *t*-test, p-value <0.05). Residual herbs had significantly higher cover in WS-2 than WS-3 throughout most years after logging (Fig. 51). Invader shrubs and herbs exhibited insignificantly higher cover in WS-2 than WS-3 throughout most years after logging. In the third and fourth years post-logging WS-3 had a significantly higher proportion of invader species to residual species than WS-2 (Fig. 52).

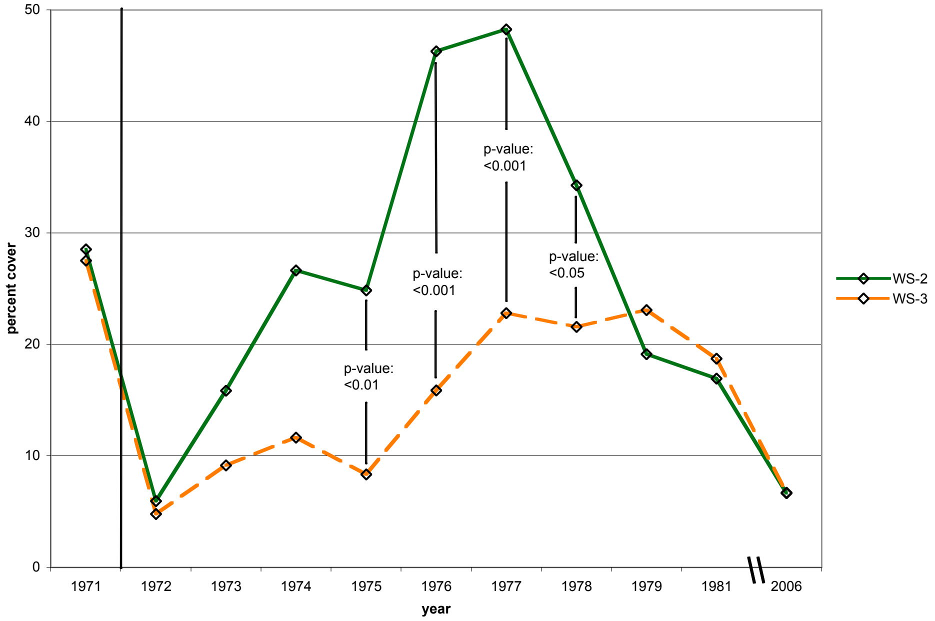

Fig. 51. Change in residual herbs in WS-2 & WS-3; plots stratified by logging treatment (n = 39 plots) (p-values from two sample *t*-test).

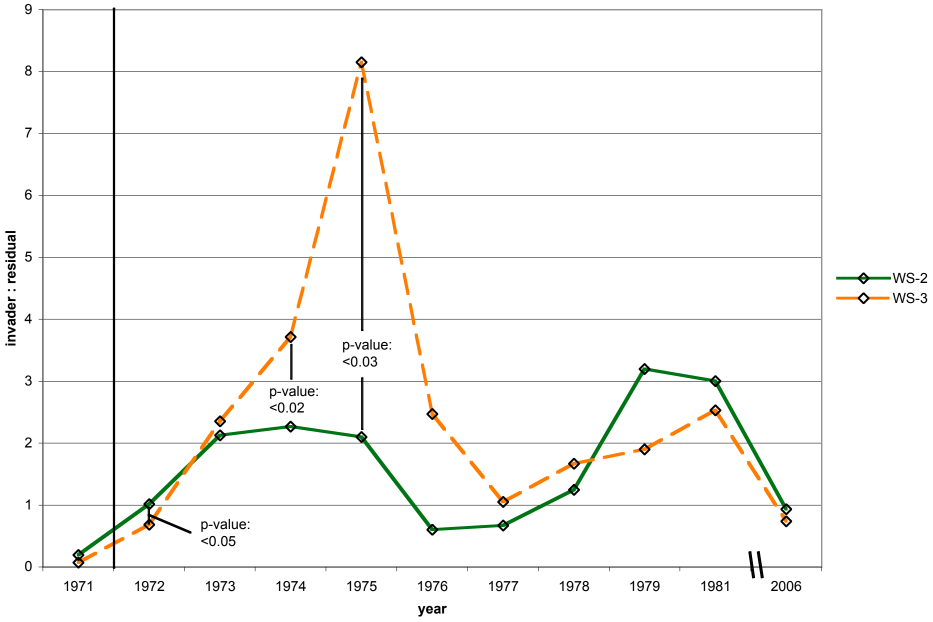

Fig. 52. Ratio between invader and total residual herbs where 1 on *y*-axis is 1 to 1 invader to residual; plots stratified by logging treatment (n = 39 plots) (p-values from two sample *t*-test).

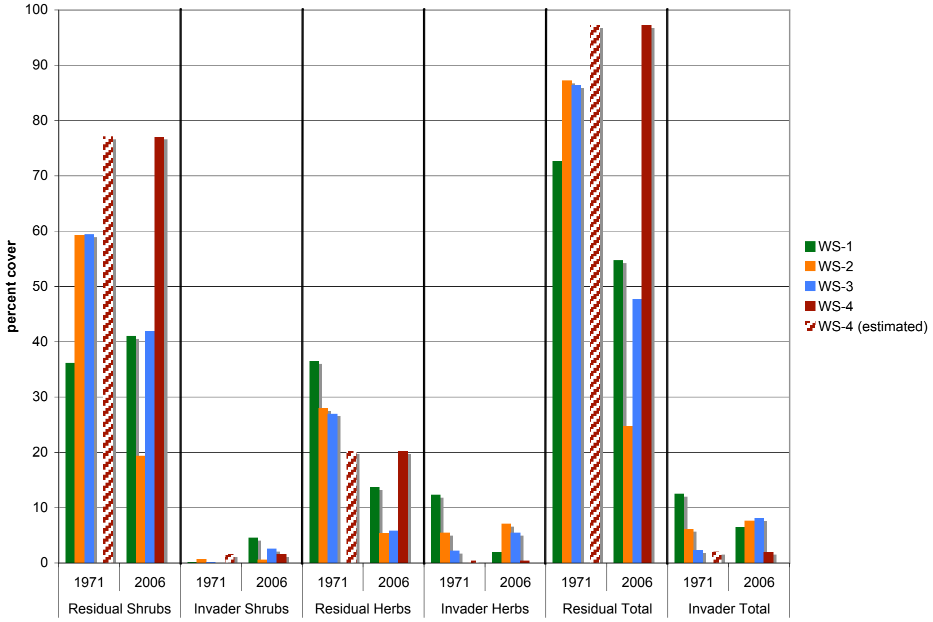

Fig. 53. Total cover in 1971 & 2006 of residual and invader shrubs and herbs of WS 1-4; plots stratified by logging treatment (1971: n = 50 plots; 2006: n = 57 plots (n = 77 plots including WS-4)); data for WS-4 in 1971 estimated from 2006 data.

Considering all three treated watersheds (1971: n = 50 plots; 2006: n = 57 plots) and the control (2006: n = 77 plots), residual species (shrub and herb) cover in 2006 was lower than in 1971 in all three treated watersheds (Figs. 53). Invader species cover was lower in 2006 than in 1971 in WS-1; however, invader species cover was higher in 2006 than in 1971 in WS-2 and WS-3. Residual shrub cover declined in both WS-2 and WS-3 between 1971 and 2006, but increased in WS-1, while residual herbs declined in the same period in all three watersheds. Invader shrub cover increased in all three watersheds, while invader herb cover increased in WS-2 and WS-3 and declined in WS-1 in the same period.

Relationship of in-watershed location to residual and invader species after clearcutting (WS-3)

Prior to logging, residual shrubs had insignificantly higher cover in center than edge plots (59.1 and 55.7 percent, respectively). Residual herbs had higher cover on edge than center plots (32.9 and 21.4 percent, respectively). Invader shrub and herb cover were higher on edge (0.1 and 3.5 percent, respectively) than center plots (0.01 and 0.2 percent, respectively).

Following logging, residual and invader shrubs and herbs had higher cover on edge than center plots; however, there were no significant differences. Additionally, at no point was the ratio of residual to invader species significantly different between edge and center plots; however, the peak in the ratio occurred on edge plots one year prior to center plots (Fig. 54).

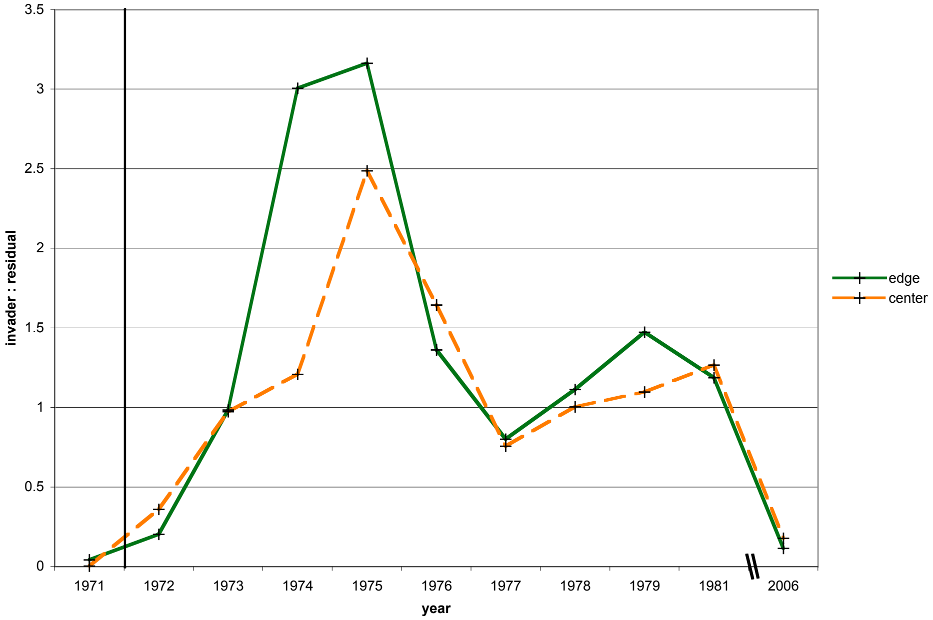

Fig. 54. Ratio between total (shrubs + herbs) invader and residual species where 1 on *y*-axis is 1 to 1 invader to residual; plots stratified by relative location (n = 18 plots).

Relationship of logging method to residual and invader species after patchcutting (WS-2)

In the pre-logging period, residual shrubs and herbs had insignificantly higher cover on plots that would be cable-yarded (53.9 and 33.2 percent, respectively) than plots that would be tractor-logged (68.6 and 24.6 percent, respectively). Conversely, invader shrubs and herbs had significantly higher cover on plots that would tractor-logged (1.5 and 9.2 percent, respectively) than plots that would be cable-yarded (0.1 and 2.6 percent respectively; two sample *t*-test <0.04 and <0.02, respectively).

Following logging, residual shrubs and herbs had consistently higher cover on cable-yarded than tractor-logged sites. The difference between the logging sites was significant most years (Fig. 55). Conversely, invader shrubs and herbs had higher cover on tractor-logged sites; however, only in the third and fourth years after logging was this difference significant among herbs and at no point was it significant among shrubs (Fig. 56).

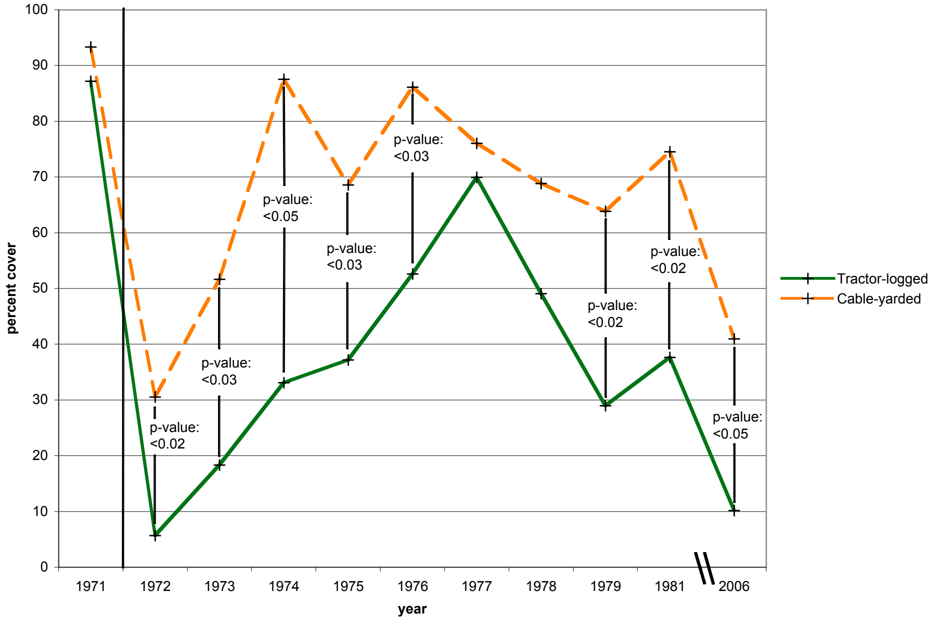

Fig. 55. Change in total (shrub + herb) residual cover in WS-2; plots stratified by logging method (n = 20 plots) (p-values from two sample *t*-test).

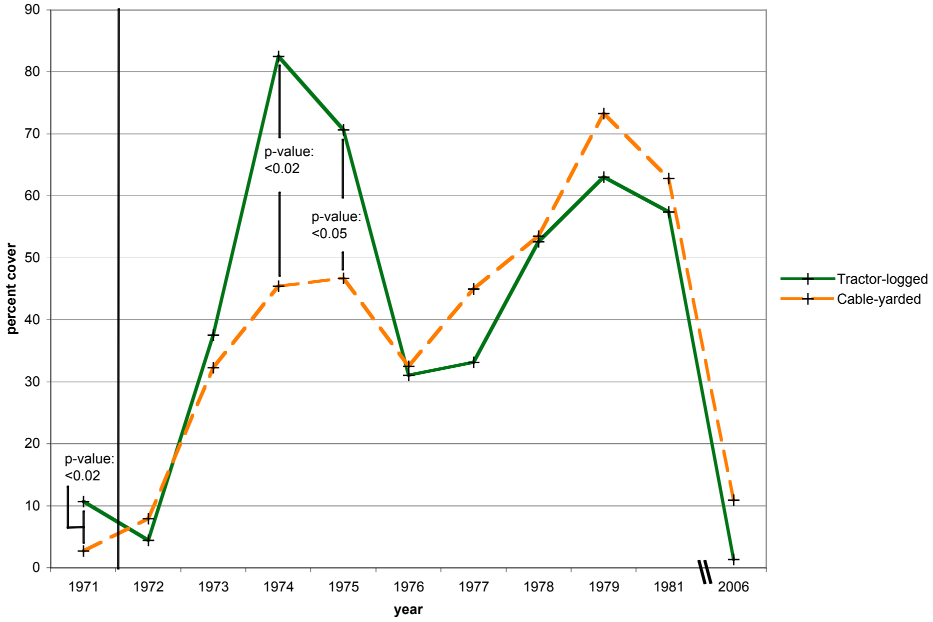

Fig. 56. Change in total (shrub + herb) invader cover in WS-2; plots stratified by logging method (n = 20 plots) (p-values from two sample *t*-test).

Relationship of soil disturbance to residual and invader species after patchcutting (WS-2) and clearcutting (WS-3)

Pre-logging data from plots stratified by post-logging soil disturbance showed residual shrubs and herbs had insignificantly higher cover on plots that would become heavily disturbed (65 and 30.2 percent, respectively) than plots that would become either lightly (58.4 and 27.1 percent, respectively) or moderately disturbed (56.8 and 27.1 percent, respectively). Invader shrub cover was similar on plots regardless of impending soil disturbance. Cover of invader herbs was higher on plots that would become heavily disturbed (5.7 percent) than plots that would become either moderately or lightly disturbed (4.1 and 2.4 percent, respectively).

For the majority of years after logging, residual shrubs had significantly higher cover on lightly disturbed sites, with its lowest cover consistently on heavily disturbed sites (ANOVA, p-value <0.05). Residual herbs had consistently higher cover on lightly disturbed sites than moderately and heavily disturbed sites. Only the first year after logging was statistically significant (ANOVA, p-value <0.05). Invader shrubs and herbs had higher cover on moderately disturbed sites for the majority of years after logging. In the second, third and tenth years after logging invader shrubs and herbs had a significantly higher proportion on heavily disturbed sites than residual shrubs and herbs on lightly disturbed sites (Fig. 57). In the eighth years post-logging moderately disturbed sites had a significantly higher proportion of invaders to residuals than lightly disturbed sites.

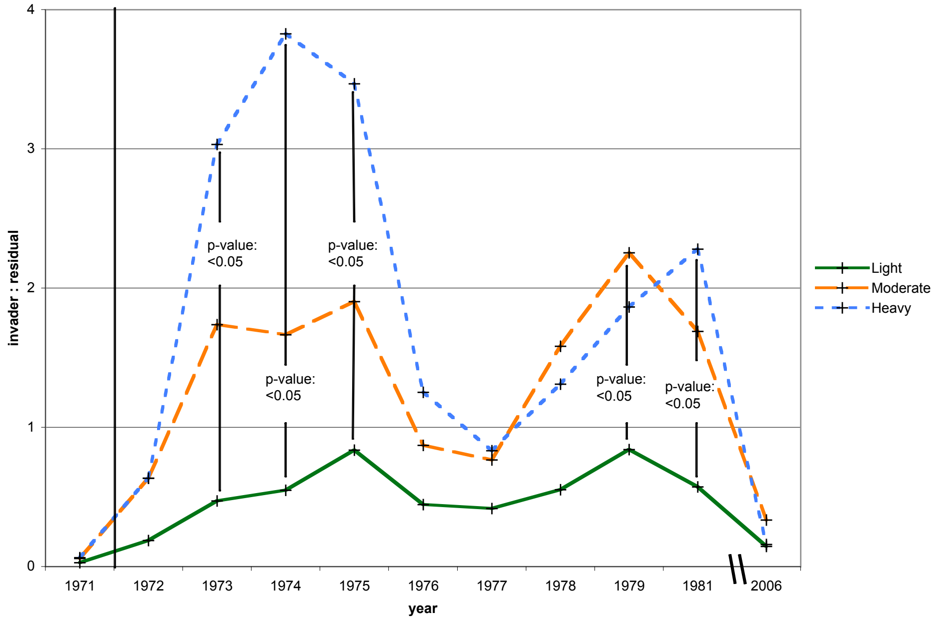

Fig. 57. Ratio between total (shrubs + herbs) invader and total residual species where 1 on *y*-axis is 1 to 1 invader to residual; plots stratified by soil disturbance (n = 36 plots) (p-values from ANOVA)

4.4 <u>Stem Counts</u>

Stem counts were completed in the field, but density estimates and watershed interpolation were not completed, therefore results are inconclusive. Maps were generated from field data (App. E) and data were deposited with the Forest Science Databank for further research.

5. Discussion

5.1 Successional Patterns of Vegetation Structures

The results indicate that environmental and disturbance variables did not affect the different vegetation groupings equally. Some variables appear to have had a greater impact on the succession of certain vegetation groupings while having little or no effect on others.

Of the eight variables tested, logging treatment and the level of soil disturbance had the most noticeable effect upon total vegetation cover (Figs. 25 and 31). In the treatment effect, the difference was primarily composed of greater shrub and herb cover (Fig. 26) in WS-2 than WS-3. This significantly higher total vegetation and herb cover may be attributed to the higher edge to area ratio of the patchcuts (WS-2) than the large clearcut (WS-3). In addition to contributions from remnant above and below ground foliage and seedbed, herbs and shrubs apparently have more readily colonized the patchcuts from the surrounding intact forest. This is supported by the edge to center analysis conducted on WS-3 cover data. Although differences were not statistically significant, both herbs and shrubs (and total vegetation) exhibited higher cover on the edges of WS-3.

These findings are in contrast to a similar study at H.J.A. where herbs exhibited peak cover in a large clearcut (WS-1) four years before three smaller patchcuts (WS-3); however in these sites there appeared to be greater species diversity and higher cover retention in the clearcut immediately following logging (Halpern 1989).

Additionally, total vegetation cover appeared to be significantly affected by the level of soil disturbance from logging methods (Fig. 31). Differences in total vegetation cover were primarily due to significantly higher shrub cover, in particular *Rubus ursinus* and *Gaultheria shallon* (Fig. 32). Rapid growth of both species is attributed to vigorous rhizomes and aerial stems that expand between the humus and soil layers (Huffman et al. 1994). Numerous studies have demonstrated a vegetative expansion of salal after tree canopy removal or thinning (Bunnell 1990; Tappeiner and Zasada 1993; Klinka et al. 1996). Similar studies in the H.J. Andrews found rapid expansion of *R. ursinus* and the persistence of *G. shallon* cover immediately following forest harvest (Dyrness 1973; Halpern 1989).

A similar result in shrub cover occurred on cable-yarded sites versus tractor-logged sites in WS-2 (Fig. 29). Cable-yarding caused less soil compaction, surface soil loss and reduced mechanical damage to the understory, leaving a significantly higher cover of *Gaultheria shallon*, *Rubus ursinus* and *Rosa gymnocarpa* on these sites (Minore 1977; Johnson & Beschta 1980). The significant post-logging differences in shrub cover on three slope classes (Fig. 22) are likely the result of greater soil disturbance caused by tractor-logging which was conducted primarily on gentler slopes (Harr et al. 1979; Johnson & Beschta 1980).

In contrast to shrubs, overstory tree species exhibited their highest cover on sites that were tractor-logged, heavily disturbed and with gradual slopes (Figs. 22 and 30). Logging disturbance probably outweighed the importance of slope on tree regeneration, as there were no significant differences among the three slope classes prior to logging.

Rather, the exposure of bare mineral soil in these sites reduced competition from shrubs and prepared the seedbed for rapid tree regeneration, as reported by Minore and others (1977) and other studies (Lanini & Radosevich 1986; Gray et al. 2005). Both Minore (1977) and Beschta and Johnson (1980) reported relatively low soil compaction in the Coyote Creek logged areas, which may account for continued overstory growth.

5.2 Relationship of Environment and Disturbance to Overstory Species

As expected, prior to disturbance, overstory species composition of each watershed appeared to be affected by relative amount of solar radiation and soil moisture (Fig. 38). Because WS-1 and WS-2 are roughly east-west trending watersheds, these areas receive higher solar energy inputs than either WS-3 or WS-4 (Fig. 5). Because of this, ponderosa and sugar pines cover was higher in WS-1 and WS-2 than WS-3 or WS-4. In contrast, WS-3 and WS-4 provided cooler, moister conditions than either WS-1 or WS-2 resulting in a higher cover of grand fir, incense cedar and western hemlock in those watersheds. The suppression of fire may have also contributed to the higher cover of shade tolerant species (Barbour et al. 2002; Smith et al. 2005), though fire studies in the watersheds have not been conducted.

Similar to the vegetation groupings results, individual overstory species appeared to be affected differently by the two sets of variables presented here. In general, disturbance variables had a more significant effect on overstory species than environmental variables. Of the environmental variables, aspect appeared to be the strongest predictor of some overstory species succession. Of the disturbance variables, only the edge-to-center analysis failed to provide statistically significant results for spatial differences in any overstory species.

The significant increase in cover experienced by grand fir in WS-1 may have been due to increased resources made available from the shelterwood tree removal. Ferguson and Adams (1980) and Seidel (1980) demonstrated significant growth in young and sapling grand fir when overstory was removed, noting that those trees under denser canopies would likely continue to grow slower than those under relatively thin canopies. Although they did not measure grand fir growth, Latham and Tappeiner (2002) demonstrated significant increases in the bole development of mature and old-growth conifers in southwestern Oregon when stand density was reduced through thinning.

It should be noted that surveys in WS-1 were conducted 35 years apart (1971 and 2006) and therefore grand fir (and other overstory species) that would have been encountered as saplings (<12.5 cm dbh) in the first survey may have been classed as mature trees in the second survey. No attempt was made to establish tree ages at the time of either survey.

Douglas-fir, ponderosa pine and incense cedar may have successfully competed for space against grand fir in WS-3, causing a sizable reduction in grand fir in the canopy (Table 10). The significant declines in cover of grand fir saplings in WS-2 and WS-3 may have been due to the closing of the canopy in both watersheds. Additionally, grand fir sapling cover may have significantly declined in WS-3 because of the decline in its prominence in the canopy in that watershed (Table 11).

Aspect appeared to have some control over the growth of grand fir in WS-2 and WS-3. Grand fir may have exhibited significantly higher cover on northwest sites because of lower surface temperatures and reduced evapotranspiration on these sites than on southeast-facing sites (Minore 1979).

By contrast, sugar pine favored southeast-facing over northerly-facing plots in WS-2 and WS-3 because of its need for high solar radiation (Minore 1970; Gray et al. 2005); however, sugar pine was not observed in the thirty-fifth year after logging irrespective of aspect position. Despite differences in aspect, logging treatment appeared to be the greatest factor controlling the distribution of sugar pine in all three watersheds. Although not significant, the increase of sugar pine in WS-1 was probably due to release of mature trees from the shelterwood cutting (Latham and Tappeiner 2002). The initial increase in light, and soil moisture and nutrient availability may have increased sugar pine sapling cover in WS-1 (Gray et al. 2005).

Sugar pine's significant decline in the WS-2 patchcuts and the WS-3 clearcut probably was apparently due to competitive stress. Stocking surveys reported in 1976 found that ponderosa pine and Douglas-fir seedlings were taller than all other overstory species (Minore 1977). Research from the Sierra Nevada found that ponderosa pine tends to grow faster and longer during the growing season than sugar pine (Lanini and Radosevich 1986). Additionally, ponderosa pine and Douglas-fir appear to have a shorter lag time between increased resource availability (i.e. shrub reduction) and growth gains from these additional resources than sugar pine (Lanini and Radosevich 1986). These concomitant effects may have resulted in overtopping and suppression of shade intolerant sugar pines in both watersheds, which may explain its absence in 2006. Prolonged shading and high stand density has been demonstrated to adversely affect sugar pine populations in California (Woodward and Land 1984; van Mantgem 2004).

Additionally, sugar pine populations have been threatened in the west by the pathogen white pine blister rust (*Cronartium ribicola* J.C. Fisch.) (Kinloch 2003; van Mantgem 2004). Surveys for the rust in Coyote Creek in 2001 resulted in positive identification of young sugar pine mortality from blister rust, further driving down their numbers in WS-2 and WS-3 (Goheen unpublished report 2001: App. D).

Conversely, ponderosa pine increased cover in WS-2 and WS-3 by taking advantage of disturbance conditions caused by logging treatments. Like other conifers, ponderosa pine seed germinates best on bare mineral soil (Minore 1977; Minore 1979; Lanini and Radosevich 1986); in addition, the removal of shrubs increases resource availability for remaining vegetation (Gray et al. 2005; Zhang et al. 2006). For these reasons, ponderosa pine had significantly higher cover on moderately to heavily disturbed and tractor-logged sites (Figs. 39 and 41). The significant difference in ponderosa pine cover between WS-2 and WS-3 may be due to stronger planted stock in WS-3 or more favorable site selection. Ponderosa pine was in greater densities on skids and decommissioned roads in WS-3 than elsewhere throughout the watersheds (personal observation); however, no attempt to quantify this affinity was made, except via the soil disturbance analysis. The insignificant ponderosa pine sapling cover increases in WS-2 and WS-3 is a result of larger numbers of ponderosa pine seed trees particularly in WS-3, while the increase in WS-1 is from favorable seedbed conditions from tractor-logging.

For the same reasons, Douglas-fir expressed significantly higher cover on similar sites in WS-2 and WS-3 (Figs. 40 and 42). The difference between WS-2 and WS-3 may be explained by the poor quality planted seedlings reported by Minore (1977) who estimated that by 1976 the majority of seedlings in both watersheds were from natural seeding. If this was the case, the intact old-growth surrounding the smaller patchcuts may have added more seed and been more effectively distributed across the logged sites in WS-2 than WS-3. The significant Douglas-fir sapling cover increase in WS-1 is attributed to exposure of bare mineral soil from tractor-logging the site. Large swaths of dense Douglas-fir sapling regeneration were observed on old skid trails in WS-1 (personal observation); however, no attempt to date or quantify the number of stems per area was made.

Logging treatment has had the most significant effect upon the occurrence western hemlock. Mature and sapling western hemlock cover increases, though not statistically significant, in WS-1 may be attributed to release. In WS-1, western hemlock saplings exhibited densities similar to that of Douglas-fir saplings. Although the stands were younger than WS-1, Bailey and Tappeiner (1998) recognized increased growth of hemlock saplings in thinned stands in western Oregon. It is possible that the temporary loss of understory vegetation, in particular *Gaultheria shallon*, may have improved site conditions for hemlock germination. *G. shallon* can negatively affect hemlock regeneration by using available nitrogen from the soil (Mallick and Prescott 2001). The significant declines of mature and sapling hemlock cover in WS-3 and low quantity in WS-2 are not surprising given the dense *G. shallon* cover in many plots as well as hemlock's site preference for (though not reliance upon) cool, moist sites and nurse logs which are essentially absent from both watersheds (Christy and Mack 1984).

The incense cedar cover increase in WS-1 is probably due to release, change in measurement status (saplings to mature) or site differences. Aspect may be the largest indicator of incense cedar recovery in WS-2 and WS-3. Cedar favored northwest-, north- and northeast-facing plots over southeast- and east-facing sites because of reduced solar intensity and concomitant increased soil moisture availability. Incense cedar has high germination and survival rates in partially shaded sites (Stark 1965) and high sapling mortality in areas receiving direct solar radiation, probably because of soil desiccation (Gray et al. 2005).

5.3 Relationship of Environment and Disturbance to Residual and Invader Species

Residual and invader species were not affected equally by the variables tested. Again, logging method and treatment, and consequent soil disturbance appeared to be the surest predictor of residual and invader composition within Coyote Creek. Aspect and slope may have had some additional influence on species colonization following logging.

Residual shrubs were more prolific in cable-yarded sites (WS-2) and lightly disturbed sites (WS-2 and WS-3) because of less damage and intact reproductive structures following logging. Early establishment in lightly disturbed sites allowed for residual shrubs to compete with invading shrub and herb species during the subsequent wave of recolonization and invasion. *Gaultheria shallon* and *Rubus leucodermis* most likely regenerated from rhizomatous growth as described by numerous investigators (Bunnell 1990; Tappeiner and Zasada 1993; Huffman et al. 1994; Klinka et al. 1996).

The persistence of *Acer circinatum* was almost certainly from remnant vegetation as well. Tappeiner and Zasada (1993) found that vine maple seedling emergence was significantly lower in clearcuts, seed predation was extremely high in clearcuts, and any emergent seedlings were preferred deer and elk browse. *Berberis nervosa* appears to have maintained a presence in the watersheds through remnant vegetation, but increased in recent years from both seed germination and rhizomes during stem initiation (Huffman and Tappeiner 1997). *Rosa gymnocarpa* persisted and occasionally increased in cover likely because of rhizomatous growth and possible additions from seedling germination (Morgan and Neuenschwander 1988A).

Overall, invader shrubs appeared to be most significantly affected by soil disturbance. However, aspect was possibly of additional importance for the growth of *Ceanothus* spp It was expected that *Ceanothus* spp would be found primarily on heavily disturbed sites because of the influence of fire on seed germination (Morgan and Neuenschwander 1988A/B); Kauffman and Martin 1991). Unfortunately, plot level disturbance surveys did not explicitly note fire intensity (if any), but it was assumed that lightly and moderately disturbed plots were not burned sites. This assumption is supported by observed persistence of *Gaultheria shallon* cover on these sites.

As expected, *Ceanothus* spp (*C. sanguineus*, *C. integerrimus* and *C. velutinus*) had higher cover on southeast- and east-facing sites where solar requirements are more readily met in Coyote Creek. Kauffman and Martin (1991) suggested that germination of *Ceanothus* spp seeds would be possible without fire if soil temperatures increased to 60°C. Comparisons demonstrate an inverse relationship between *Ceanothus* spp cover and *G. shallon*/*R. ursinus* cover. Therefore, sites with little or no remnant vegetation may have received the direct solar energy necessary to heat the soil to temperatures sufficiently for *Ceanothus* spp seed germination. To be sure, *Ceanothus* spp likely germinated and increased cover on heavily disturbed and burnt over sites as well.

Found sporadically in the mature forest understory, *Symphoricarpos mollis* had very low cover on heavily disturbed sites probably because of mechanical or fire damage to rhizomes. Where *S. mollis* was found on these sites, it might have been from germinated seed (Morgan and Neuenschwander 1988B). By contrast, *Rubus leucodermis* had much higher cover on heavily disturbed sites. The exposure of bare mineral soil provides a favorable seedbed for local raspberry seed germination (Morgan and Neuenschwander 1988A/B).

Residual herbs appear to be most significantly affected by logging treatment and soil disturbance (Fig. 51). As previously discussed, the close proximity of the intact forest edge probably contributed to the reseeding of the small patchcuts in WS-2. Although the edge to center analysis resulted in insignificant differences in residual herb cover, the plots in WS-2 were, in most cases, much closer to the edge than those in WS-3. Additionally, the sites in WS-2 generally were on the same slope, aspect and soil series as the surrounding stand. Similar to residual shrubs, perennials such as *Xerephyllum tenax*, *Linnaea borealis* and *Whipplea modesta* had little recolonization from seed in heavily disturbed sites, in contrast to increased growth from rhizomes and stolons in lightly disturbed sites (Antos and Halpern 1997).

Invader herbs may have initially benefited from the reduced residual shrub and herb cover on moderately and heavily disturbed sites as well as the higher solar radiation on southeast-facing sites (Fig. 49). Dyrness (1973) found *Senecio sylvaticus* had higher cover on burned over sites. West and Chilcote (1967) suggested *S. sylvaticus* had higher cover on burnt sites because of high nitrogen requirements readily available in fire affected soils and reduced competition. In Coyote Creek, *S. sylvaticus* did not appear to favor any of the disturbance levels, although fire intensity at the plot level unfortunately was not measured in disturbance surveys.

However, *S. sylvaticus* was higher on east- and southeast-facing sites, suggesting possible solar requirements better met by these aspects than on northerly-facing sites. It is unlikely that the sudden decline in *S. sylvaticus* observed in Coyote Creek and elsewhere (see West and Chilcote 1968; Dyrness 1973; Schoonmaker and McKee 1988; Halpern et al. 1997) in the third year following logging was due to reduction in light intensity. Halpern and others (1997) demonstrated that interspecific competition does not cause the sudden decline in *S. sylvaticus*, but suggested that reductions of soil nutrients, particularly potassium, may be important. Unfortunately, the results presented here do not further illuminate this phenomenon.

Both soil disturbance and aspect appeared to affect abundance of *Epilobium paniculatum*. As with *S. sylvaticus*, southeast-facing sites probably provided more solar energy than northerly-facing sites. Additionally, the initial reduction in abundance of residual shrubs and herbs (on moderately and heavily disturbed sites) may have provided space for *E. paniculatum* to germinate and reproduce. The introduction of other invader species and increase of residual species may have provided competitive pressure on *E. paniculatum* on these sites, leading to their decline. A positive association between an increase in *E. paniculatum* and bare ground (Rozell 2003), as well as individual *E. paniculatum* plant size reductions following in-growth have been previously observed (Antos and Halpern 1997).

It is likely that the intolerant *P. aquilinum* colonized lightly disturbed sites because of the increasing cover of tall shrubs (e.g. *Ceanothus* spp) and trees (e.g. Douglas-fir and ponderosa pine) on moderately and heavily disturbed sites. Puettmann and Berger (2006) observed a similar pattern of *P. aquilinum* having its highest densities in areas with fewer trees.

5.4 Relationship among Trees, Shrubs and Herbs

Overall, disturbance factors appear to have more influence than environmental characteristics on vegetation classes within these watersheds. Of the environmental characteristics observed, aspect appeared to have the most significant control over species distribution. Heavily disturbed sites had significantly more increases on plant cover because of greater mechanical damage (i.e. road building, landings, skidding) to the local understory particularly the residual shrub layer. A comparatively reduced residual shrub layer appears to have allowed invader species to take fuller advantage of the open space. Once established, invader species successfully competed with residual species on these sites for at least the first ten years following logging. In addition to the reduction in residual foliage, roads and skid trails may have provided pathways for invading species, particularly exotic herbs such as *Hypericum perforatum*, *Plantago major* and *Rumex acetosella*.

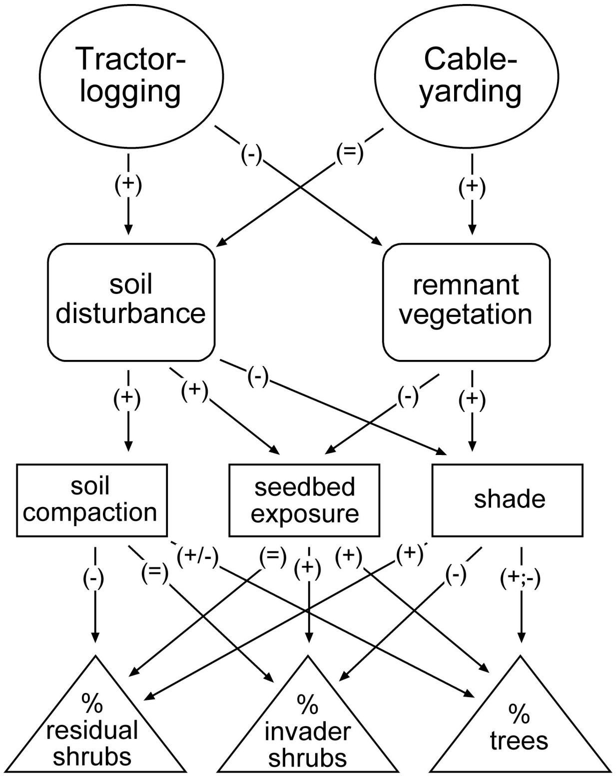

Fig. 58. Conceptual model expressing the primary driving mechanisms of competition and succession of residual shrubs, invader shrubs and overstory species; (+) is positive effect; (-) is negative effect; (=) is neutral effect; (+/-) is initial positive and subsequent negative effect; (+;-) is species dependent effect

The exposure of bare, mineral soil from logging disturbance provided favorable sites for Douglas-fir and ponderosa pine seed germination, the overstory species with the highest cover percentages in WS-2 and WS-3 (Fig. 58). Heavily disturbed sites generally lacked residual shrub and herb cover immediately following logging, thus reducing competition and shade for emergent seedlings. Nitrogen-fixing species (e.g. *Ceanothus* spp, *Vicia americana*, *Lathyrus polyphyllus*, *Lotus crassifolius*) were predominantly in the invader herb group and largely invaded the same sites as overstory species. Additional nitrogen inputs from these species may have increased the vigor and survival of overstory saplings (Binkley et al. 1982), while buffering soil moisture loss (Gray et al. 2005). Dense stands of *Ceanothus* spp and other shrubs and herbs can become a limiting factor for Douglas-fir saplings, particularly in water stressed environments (Monleon et al. 1999; Oakley et al. 2006). However, the dense overstory tree cover present today on heavily and moderately disturbed sites that once supported the highest observed cover of *Ceanothus* spp suggests that invader shrub competition was not a factor after saplings were established. In contrast, sites with dense remnant *G. shallon* and *R. ursinus* cover immediately following logging continue to support significantly lower covers of Douglas-fir and ponderosa pine today. Further, the ericaceous shrub *G. shallon* has been observed to deplete nitrogen and increase acidity to levels that suppress western hemlock and western red cedar (Mallick and Prescott 2001).

5.5 Potential Management Outcomes

The treated stands in WS-2 and WS-3 are currently in a stage of canopy closure, characterized by a dense overstory canopy severely restricting light to the understory, whereas WS-1 and WS-4 maintain vertical and horizontal diversity (Franklin et al. 2002). Because of plantation-style management and no pre-commercial or commercial thinning, as would be expected in a managed landscape, these stands have closed their canopies. As a consequence, understory vegetation cover and diversity is currently lower than existed before logging, while tree cover is much higher (Figs. 59-62).

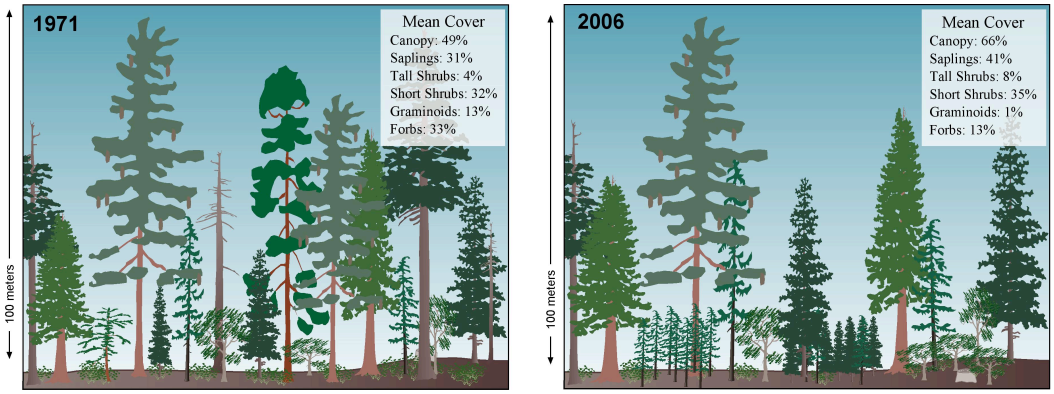

Fig. 59. Profile of vertical and horizontal structure of shelterwood thinning in WS-1 in 1971 & 2006

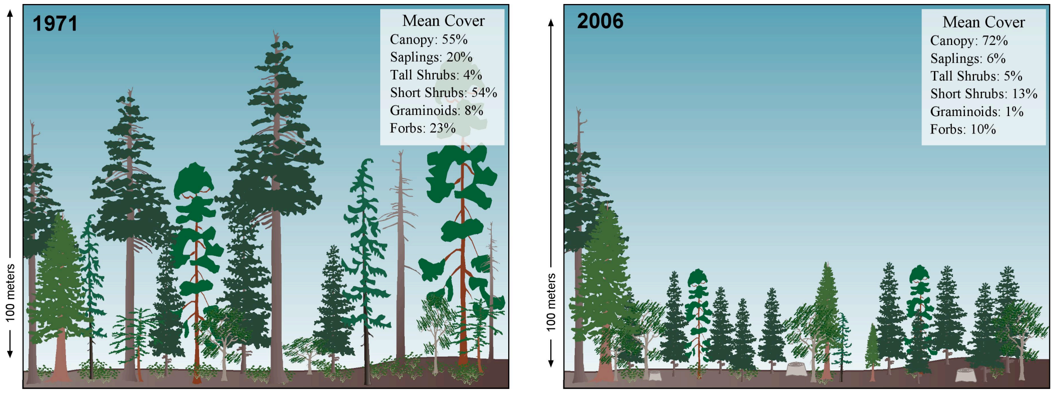

Fig. 60. Profile of vertical and horizontal structure of patchcutting in WS-2 in 1971 & 2006

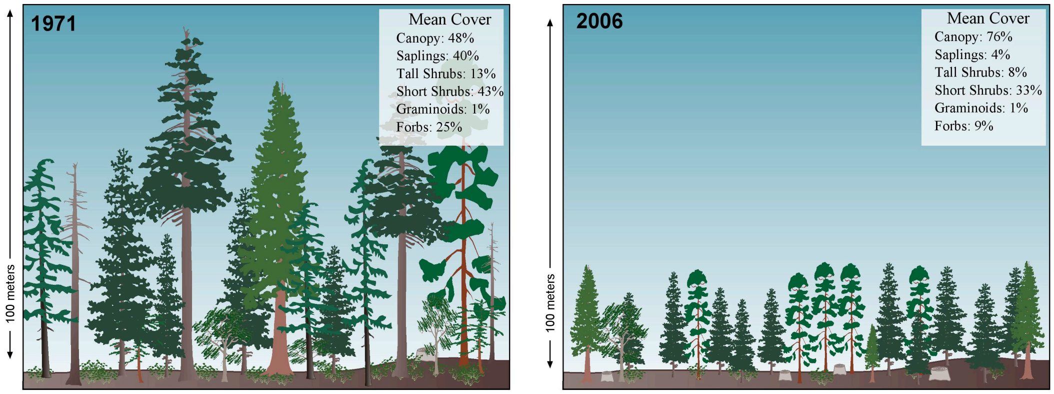

Fig. 61. Profile of vertical and horizontal structure of clearcutting in WS-3 in 1971 & 2006

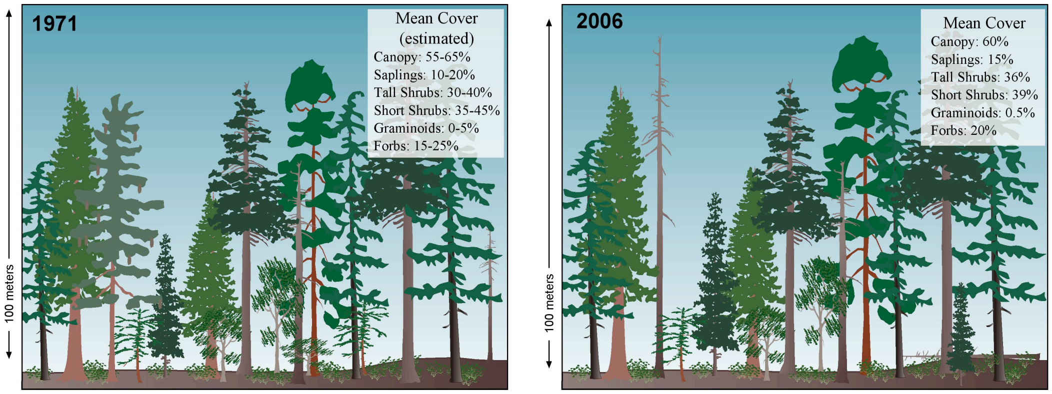

Fig. 62. Profile of vertical and horizontal structure of WS-4 control in 1971 & 2006; 1971 estimated from 2006 data with little change assumed

The conversion of mature and old-growth forests to plantations poses serious questions for land managers charged with multiple use mandates. The US Forest Service estimates 70 percent of forested lands in the Pacific Northwest are less than 100 years old (USFS 2002 from Lindh and Muir 2004). Research and management plans are currently underway to address various thinning regimes of the numerous plantation stands on public lands similar to those of the patchcuts of WS-2 and the clearcut of WS-3. In general, thinning will increase the light to the forest floor, initially increase availability of water and nutrient resources and alter habitat for wildlife (Hayes et al. 1997). However, the method, magnitude and pattern of thinning in these watersheds will have variable effects on the stand structures.

Even thinning of low magnitude (<25 percent basal area removal) with little damage to the understory vegetation would maintain horizontal homogeneity and likely result in a dominant response by shrubs, particularly *G. shallon* and *R. ursinus* (Bunell 1990; Huffman et al. 1994). Without shrub reductions, tree regeneration would probably be quite low even among shade tolerant species such as grand fir and incense cedar (Brandeis et al. 2001). Sugar pine and ponderosa pine would most likely be excluded from regeneration because of continued low light, and western hemlock may have trouble establishing if *G. shallon* maintained high densities (Christy and Mack 1984; Mallick and Prescott 2001; Gray et al. 2005). Herb response would probably be insignificant compared to the positive shrub response.

By contrast, uneven thinning of relatively high intensity (30-50 percent basal area removal) with moderate mechanical damage to the understory (removal of residual shrubs, exposure of mineral soil) would likely result in positive responses from each structural unit, particularly trees and herbs and decrease horizontal homogeneity. Exposure of mineral soil would be essential for Douglas-fir and ponderosa pine germination (Minore 1979; Lanini and Radosevich 1986) and the reduction of shrubs would aid in sapling survival (Bailey and Tappeiner 1998; Brandeis et al. 2001; Mallick and Prescott 2001). The opening of gaps of various sizes might simulate natural gaps that help create horizontal and vertical diversity in mature and old-growth stands (Spies et al. 1990; Tappeiner et al. 1997). Gaps could be sites for herb establishment and renewal; however, invaders and exotics may become persistent. Halpern and others (1999) working in the Olympic Peninsula found that forest floor litter contained much higher concentrations of seeds of exotic invaders than soil. Consequently, mineral soil exposure may favor native herbs whereas intact remnant litter may favor exotic and wind-dispersed annuals (Halpern et al. 1999).

Of particular concern for management is the health of the sugar pine population in the region; therefore, thinning treatments must account for the threats to this population. The patchcuts of WS-2 and clearcut of WS-3 had no recorded canopy or sapling sugar pines, whereas sugar pine cover increased in the shelterwood thin of WS-1. Conversion of old-growth to younger, plantation style forests has clearly changed the structure of stands across the landscape, negatively affecting sugar pine establishment (van Mantgem et al. 2004; Smith et al. 2005).

Shade tolerant species cover was higher than sugar and ponderosa pine cover in the control (WS-4). Because there is no long-term data, it is difficult to determine changes in sapling species composition; however, data from old-growth and mature stands in the Sierra Nevada suggest that fire suppression is leading to increased recruitment among shade tolerant species (van Mantgem 2004; Gray et al. 2005). Additionally, increased fuel loads from fire suppression threaten populations with more severe fires (Agee 1993).

Thinning regimes simulating fire may benefit threatened populations of sugar pine. Though not significant, the increase in sugar pine in WS-1 as well as evidence from controlled burning and thinning experiments in the Sierra Nevada promise potentially positive results for sugar pine populations (van Mantgem 2004). However, treatments to stands need to consider competitive interactions that may hinder sugar pine establishment. The increase in sugar pine sapling cover in WS-1 was accompanied by a significantly larger increase in western hemlock and Douglas-fir sapling cover. Tractor logging may have contributed to this increase, particularly of Douglas-fir because of the exposure of bare mineral soil. Leaving some slash may benefit understory species (Thomas et al. 1999) as well as favor sugar pine over ponderosa pine and Douglas-fir, which require clear site preparation (Minore 1979; Lanini and Radosevich 1986).

Providing large tracts of exposed mineral soil may lead to favorable ponderosa pine and Douglas-fir rather than sugar pine regeneration as evidenced by the near total dominance of these species in the patchcuts of WS-2 and clearcut of WS-3, particularly along decommissioned roadways and skids.

Understory regrowth may be an additional source of competition for sugar pine establishment. Lindh and Muir (2004) found that thinned young stands had higher low shrub cover (i.e. *Gaultheria shallon* and *Berberis nervosa*) than old-growth stands. As demonstrated in the patchcuts of WS-2 and clearcut of WS-3 and elsewhere (Klinka et al. 1996; Mallick and Prescott 20001; Gray et al. 2005; Zhang et al. 2006), cover of shrubs and overstory species are inversely related due to competition between the structures during early succession. Without successful removal of shrubs from the understory, sugar pine may have difficulty establishing.

6. Conclusion

Because of the long-term vegetation and streamflow records, published reports and renewed interest in experimental designs and research, the South Umpqua Experimental Forest (SUEF) is an excellent site for continued investigation of ecosystem processes and human disturbance. To meet this, the 77 vegetation survey plots will be entered into the Pacific Northwest Permanent Plot System for continued monitoring.

A number of research projects could emerge from the permanent vegetation plots in the SUEF. Plot sampling should occur before and following any second-round treatments applied to the watersheds. Silvicultural experiments designed to address the pressures from competition and white pine blister rust facing sugar pines should be conducted in the SUEF. Modeling changes in climate and increased fire risk from changing forest structure and increased drought potential could be incorporated into these experimental designs.

Continued collection of stem data including mortality should be conducted within the plots as well. These data can be added to continued streamflow data collection and analysis to assess the effects of the various harvest treatments on streamflow. Additionally, these data can document changes in overstory species composition caused by harvesting, climate change or fire suppression, and help guide future experimental designs within the SUEF as well as management decisions within southwest Oregon and northwest California.

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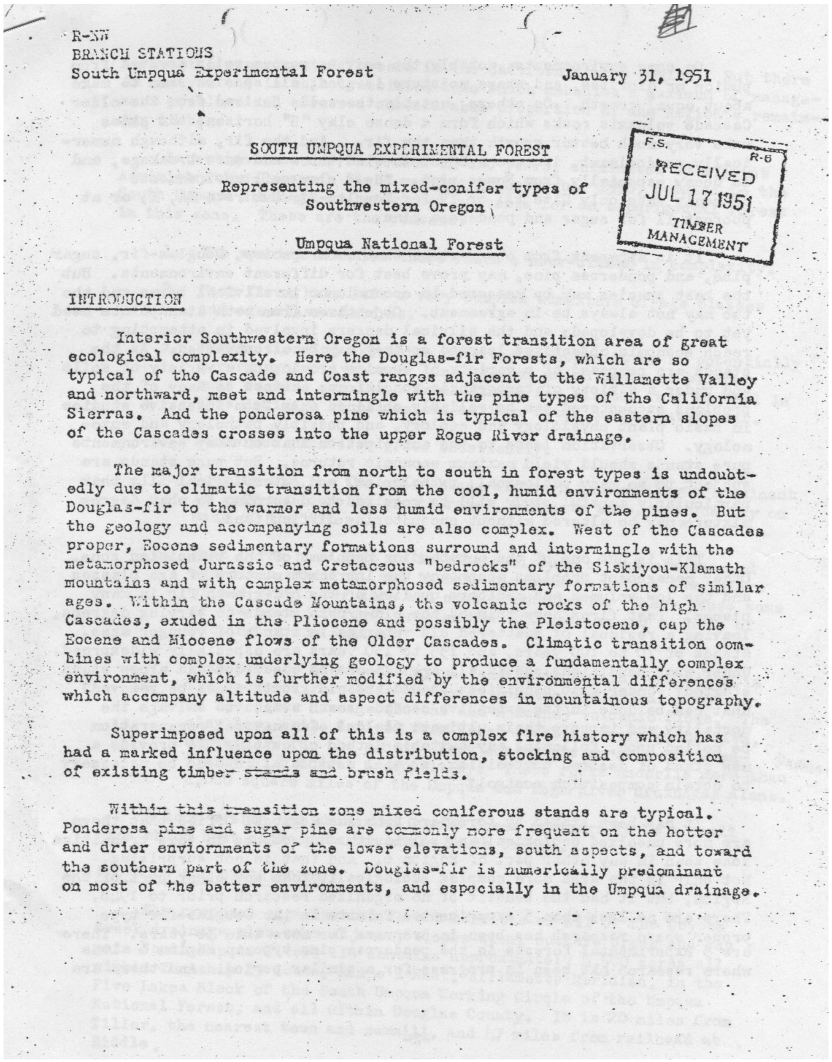
8. Appendices

Appendix A. Historical Documents

Historical documents were reviewed to capture the history of the South Umpqua Experimental Forest. The following includes samples of Forest Service memos pertaining to the official designation and the original study plan for experimental treatments. Field notes and maps have been added as well.



R-NW
BRANCH STATIONS
South Umpqua Experimental Forest

January 31, 1951

F.S. R-6
RECEIVED
JUL 17 1951
TIMBER
MANAGEMENT

SOUTH UMPQUA EXPERIMENTAL FOREST

Representing the mixed-conifer types of
Southwestern Oregon

Umpqua National Forest

INTRODUCTION

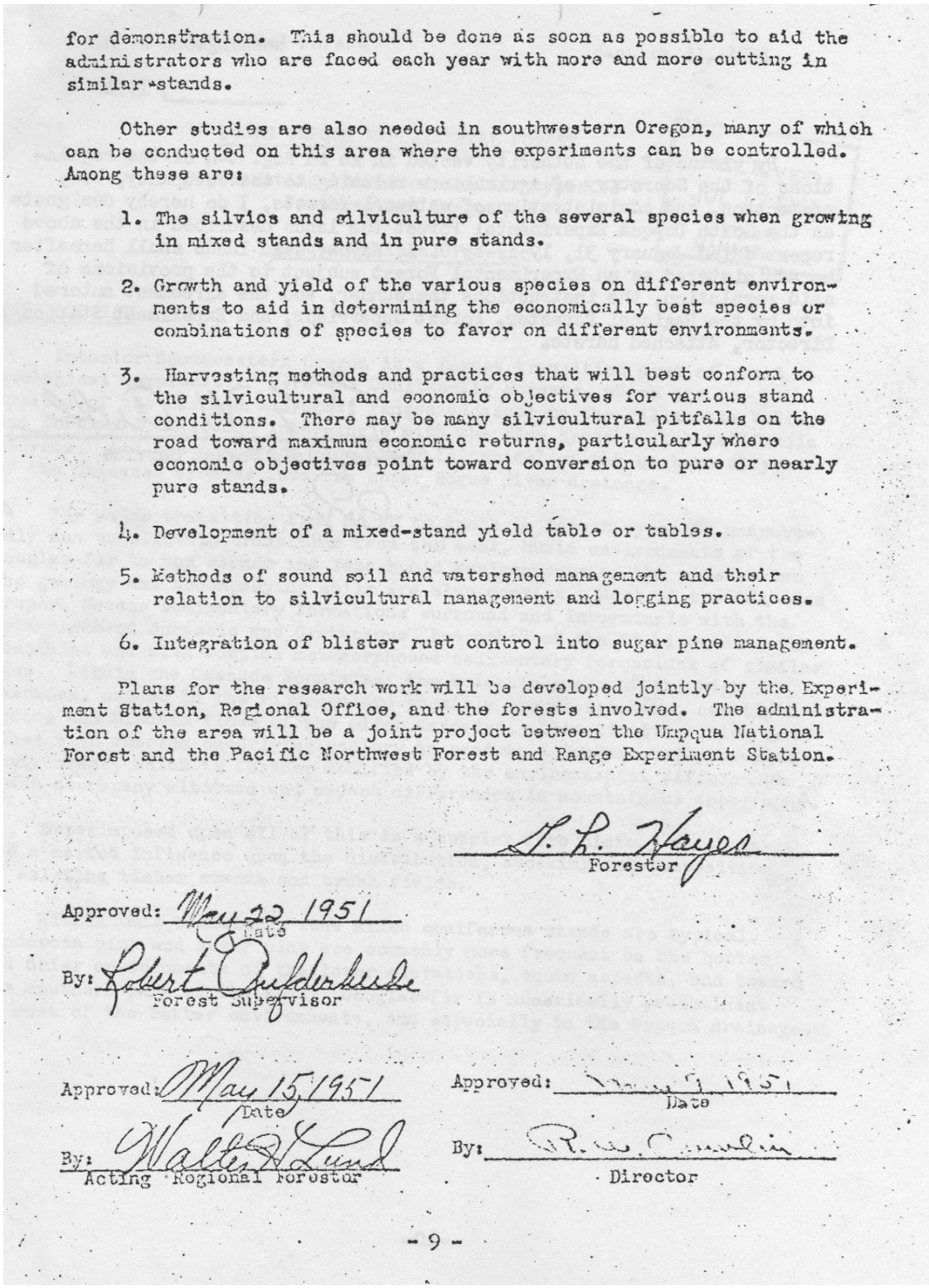
Interior Southwestern Oregon is a forest transition area of great ecological complexity. Here the Douglas-fir Forests, which are so typical of the Cascade and Coast ranges adjacent to the Willamette Valley and northward, meet and intermingle with the pine types of the California Sierras. And the ponderosa pine which is typical of the eastern slopes of the Cascades crosses into the upper Rogue River drainage.

The major transition from north to south in forest types is undoubtedly due to climatic transition from the cool, humid environments of the Douglas-fir to the warmer and less humid environments of the pines. But the geology and accompanying soils are also complex. West of the Cascades proper, Eocene sedimentary formations surround and intermingle with the metamorphosed Jurassic and Cretaceous "bedrocks" of the Siskiyou-Klamath mountains and with complex metamorphosed sedimentary formations of similar ages. Within the Cascade Mountains, the volcanic rocks of the high Cascades, exuded in the Pliocene and possibly the Pleistocene, cap the Eocene and Miocene flows of the Older Cascades. Climatic transition combines with complex underlying geology to produce a fundamentally complex environment, which is further modified by the environmental differences which accompany altitude and aspect differences in mountainous topography.

Superimposed upon all of this is a complex fire history which has had a marked influence upon the distribution, stocking and composition of existing timber ~~stands and~~ brush fields.

~~Within this transition~~ zone mixed coniferous stands are typical. Ponderosa pine and sugar pine are commonly more frequent on the hotter and drier enviornments of the lower elevations, south aspects, and toward the southern part of the zone. Douglas-fir is numerically predominant on most of the better environments, and especially in the Umpqua drainage.

Fig. 1. Introduction to the official designation of the South Umpqua Experimental Forest (1951).



for demonstration. This should be done as soon as possible to aid the administrators who are faced each year with more and more cutting in similar stands.

Other studies are also needed in southwestern Oregon, many of which can be conducted on this area where the experiments can be controlled. Among these are:

1. The silvics and silviculture of the several species when growing in mixed stands and in pure stands.
2. Growth and yield of the various species on different environments to aid in determining the economically best species or combinations of species to favor on different environments.
3. Harvesting methods and practices that will best conform to the silvicultural and economic objectives for various stand conditions. There may be many silvicultural pitfalls on the road toward maximum economic returns, particularly where economic objectives point toward conversion to pure or nearly pure stands.
4. Development of a mixed-stand yield table or tables.
5. Methods of sound soil and watershed management and their relation to silvicultural management and logging practices.
6. Integration of blister rust control into sugar pine management.

Plans for the research work will be developed jointly by the Experiment Station, Regional Office, and the forests involved. The administration of the area will be a joint project between the Umpqua National Forest and the Pacific Northwest Forest and Range Experiment Station.

S. L. Hayes
Forester

Approved: May 22, 1951
Date

By: Robert Aufderheide
Forest Supervisor

Approved: May 15, 1951
Date

By: Walter H. Lund
Acting Regional Forester

Approved: [illegible] 1951
Date

By: R. W. Cowlin
Director

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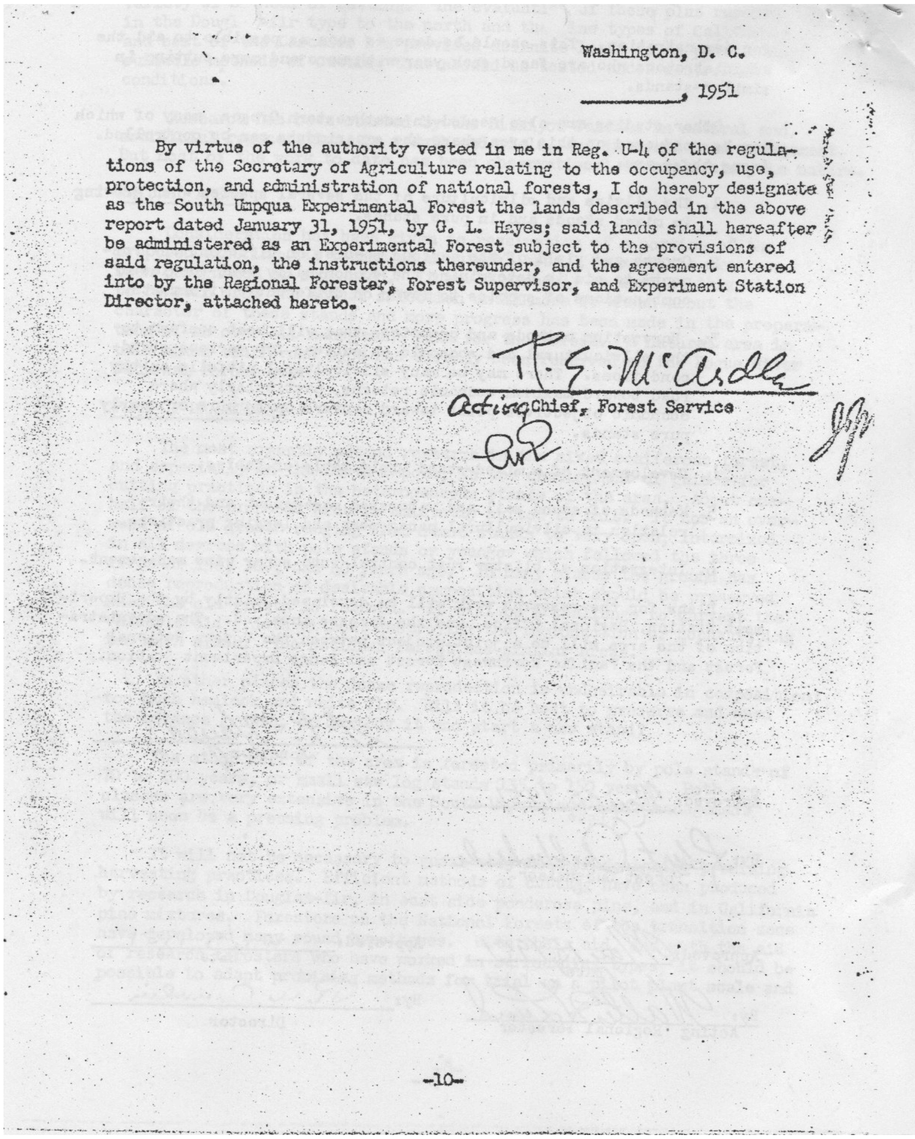
Fig. 2. Signatories to the official designation of the South Umpqua Experimental Forest (1951).



Washington, D. C.

__________, 1951

By virtue of the authority vested in me in Reg. U-4 of the regulations of the Secretary of Agriculture relating to the occupancy, use, protection, and administration of national forests, I do hereby designate as the South Umpqua Experimental Forest the lands described in the above report dated January 31, 1951, by G. L. Hayes; said lands shall hereafter be administered as an Experimental Forest subject to the provisions of said regulation, the instructions thereunder, and the agreement entered into by the Regional Forester, Forest Supervisor, and Experiment Station Director, attached hereto.

T. E. McArdle

Acting Chief, Forest Service

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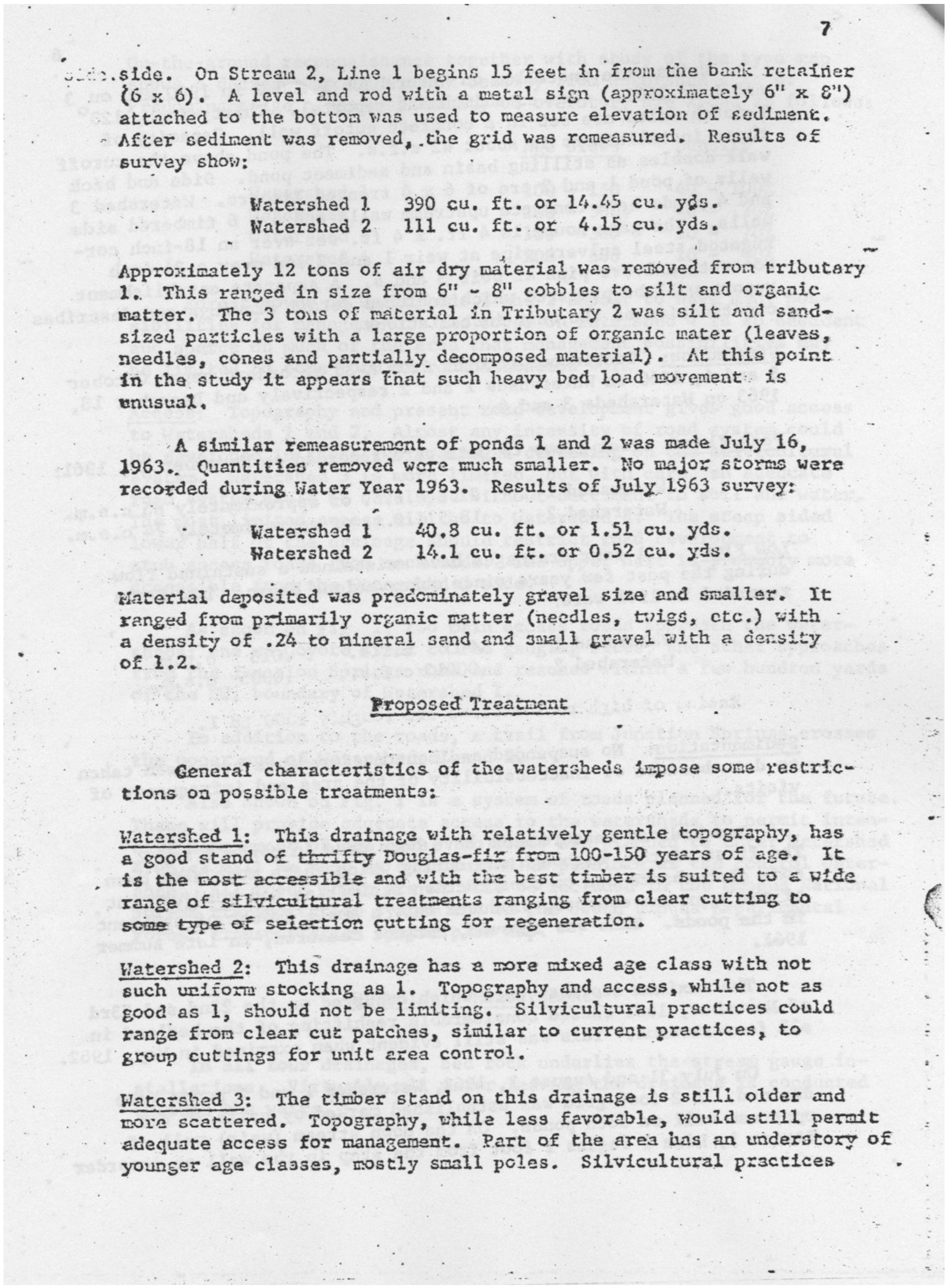
Fig. 3. Final signatory to the official designation of the South Umpqua Experimental Forest (1951).



7

side. On Stream 2, Line 1 begins 15 feet in from the bank retainer (6 x 6). A level and rod with a metal sign (approximately 6" x 8") attached to the bottom was used to measure elevation of sediment. After sediment was removed, the grid was remeasured. Results of survey show:

Watershed 1	390 cu. ft. or 14.45 cu. yds.
Watershed 2	111 cu. ft. or 4.15 cu. yds.

Approximately 12 tons of air dry material was removed from tributary 1. This ranged in size from 6" - 8" cobbles to silt and organic matter. The 3 tons of material in Tributary 2 was silt and sand-sized particles with a large proportion of organic mater (leaves, needles, cones and partially decomposed material). At this point in the study it appears that such heavy bed load movement is unusual.

A similar remeasurement of ponds 1 and 2 was made July 16, 1963. Quantities removed were much smaller. No major storms were recorded during Water Year 1963. Results of July 1963 survey:

Watershed 1	40.8 cu. ft. or 1.51 cu. yds.
Watershed 2	14.1 cu. ft. or 0.52 cu. yds.

Material deposited was predominately gravel size and smaller. It ranged from primarily organic matter (needles, twigs, etc.) with a density of .24 to mineral sand and small gravel with a density of 1.2.

Proposed Treatment

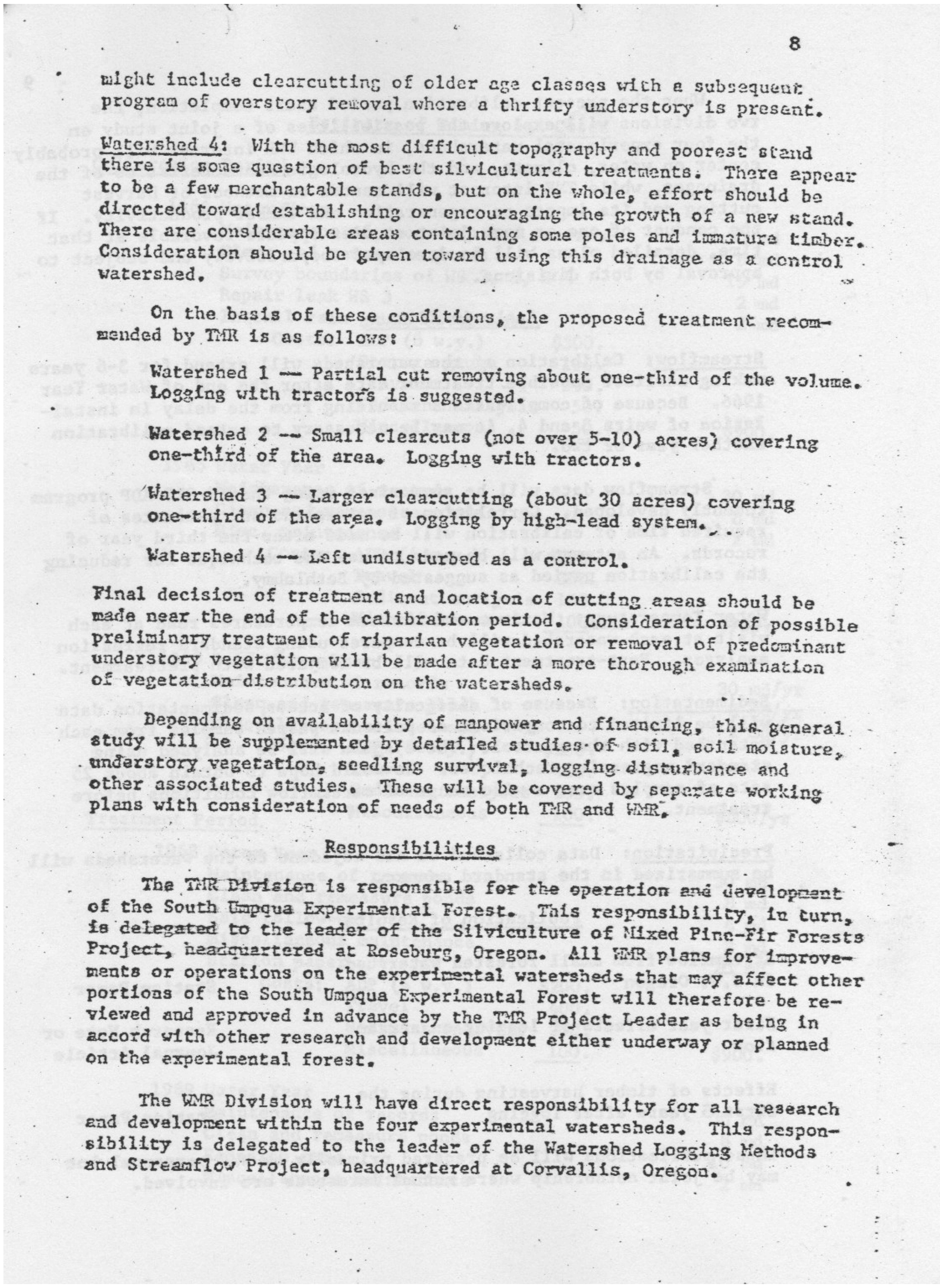
General characteristics of the watersheds impose some restrictions on possible treatments:

Watershed 1: This drainage with relatively gentle topography, has a good stand of thrifty Douglas-fir from 100-150 years of age. It is the most accessible and with the best timber is suited to a wide range of silvicultural treatments ranging from clear cutting to some type of selection cutting for regeneration.

Watershed 2: This drainage has a more mixed age class with not such uniform stocking as 1. Topography and access, while not as good as 1, should not be limiting. Silvicultural practices could range from clear cut patches, similar to current practices, to group cuttings for unit area control.

Watershed 3: The timber stand on this drainage is still older and more scattered. Topography, while less favorable, would still permit adequate access for management. Part of the area has an understory of younger age classes, mostly small poles. Silvicultural practices

Fig. 4. Page 7 of the final study plan with description and proposed treatment of watersheds (1969).



might include clearcutting of older age classes with a subsequent program of overstory removal where a thrifty understory is present.

Watershed 4: With the most difficult topography and poorest stand there is some question of best silvicultural treatments. There appear to be a few merchantable stands, but on the whole, effort should be directed toward establishing or encouraging the growth of a new stand. There are considerable areas containing some poles and immature timber. Consideration should be given toward using this drainage as a control watershed.

On the basis of these conditions, the proposed treatment recommended by TMR is as follows:

Watershed 1 — Partial cut removing about one-third of the volume. Logging with tractors is suggested.

Watershed 2 — Small clearcuts (not over 5-10) acres) covering one-third of the area. Logging with tractors.

Watershed 3 — Larger clearcutting (about 30 acres) covering one-third of the area. Logging by high-lead system.

Watershed 4 — Left undisturbed as a control.

Final decision of treatment and location of cutting areas should be made near the end of the calibration period. Consideration of possible preliminary treatment of riparian vegetation or removal of predominant understory vegetation will be made after a more thorough examination of vegetation distribution on the watersheds.

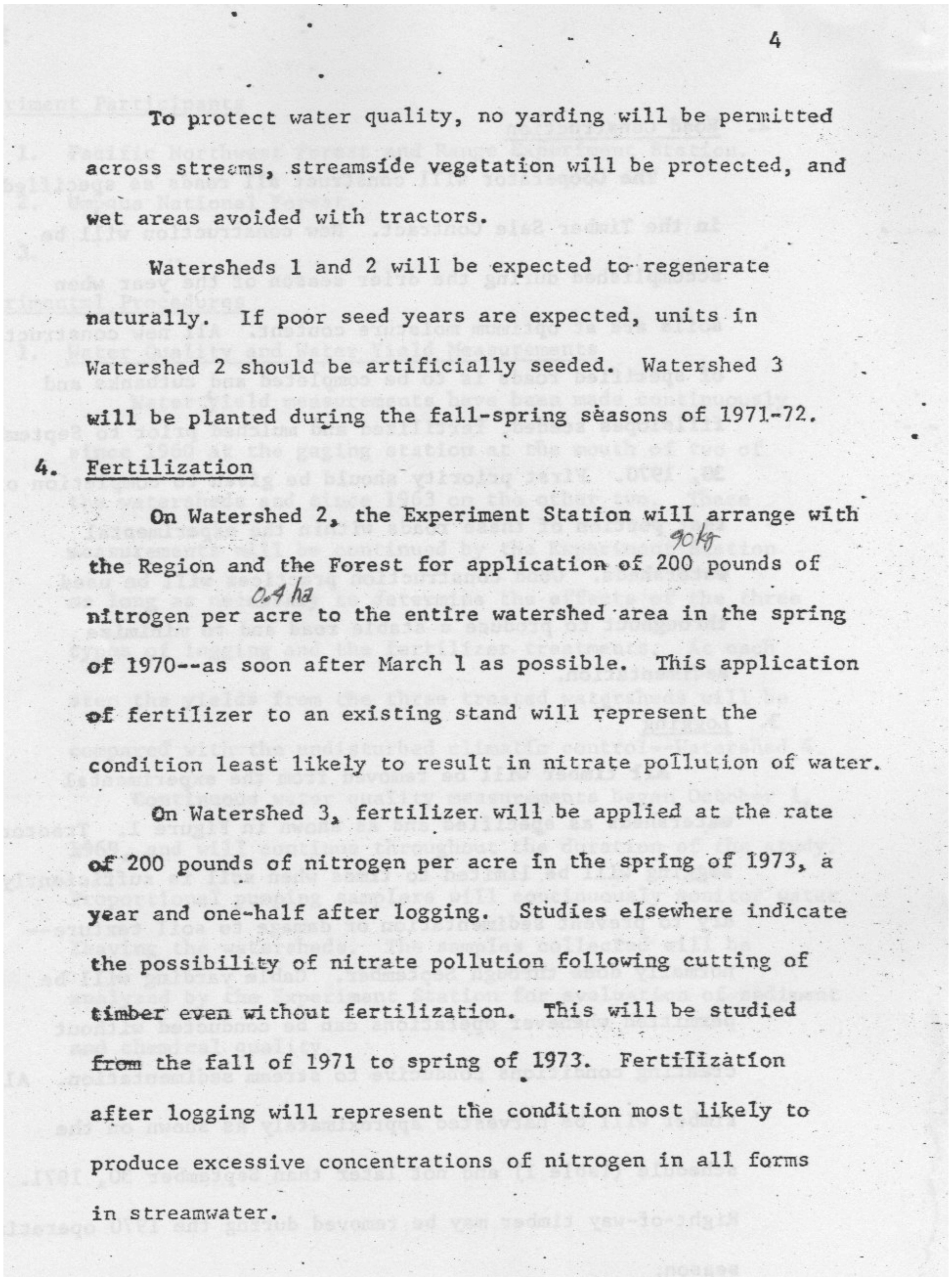
Depending on availability of manpower and financing, this general study will be supplemented by detailed studies of soil, soil moisture, understory vegetation, seedling survival, logging disturbance and other associated studies. These will be covered by separate working plans with consideration of needs of both TMR and WMR.

Responsibilities

The TMR Division is responsible for the operation and development of the South Umpqua Experimental Forest. This responsibility, in turn, is delegated to the leader of the Silviculture of Mixed Pine-Fir Forests Project, headquartered at Roseburg, Oregon. All WMR plans for improvements or operations on the experimental watersheds that may affect other portions of the South Umpqua Experimental Forest will therefore be reviewed and approved in advance by the TMR Project Leader as being in accord with other research and development either underway or planned on the experimental forest.

The WMR Division will have direct responsibility for all research and development within the four experimental watersheds. This responsibility is delegated to the leader of the Watershed Logging Methods and Streamflow Project, headquartered at Corvallis, Oregon.

Fig. 5. Page 8 of the final study plan with description and proposed treatment of watersheds (1969).



To protect water quality, no yarding will be permitted across streams, streamside vegetation will be protected, and wet areas avoided with tractors.

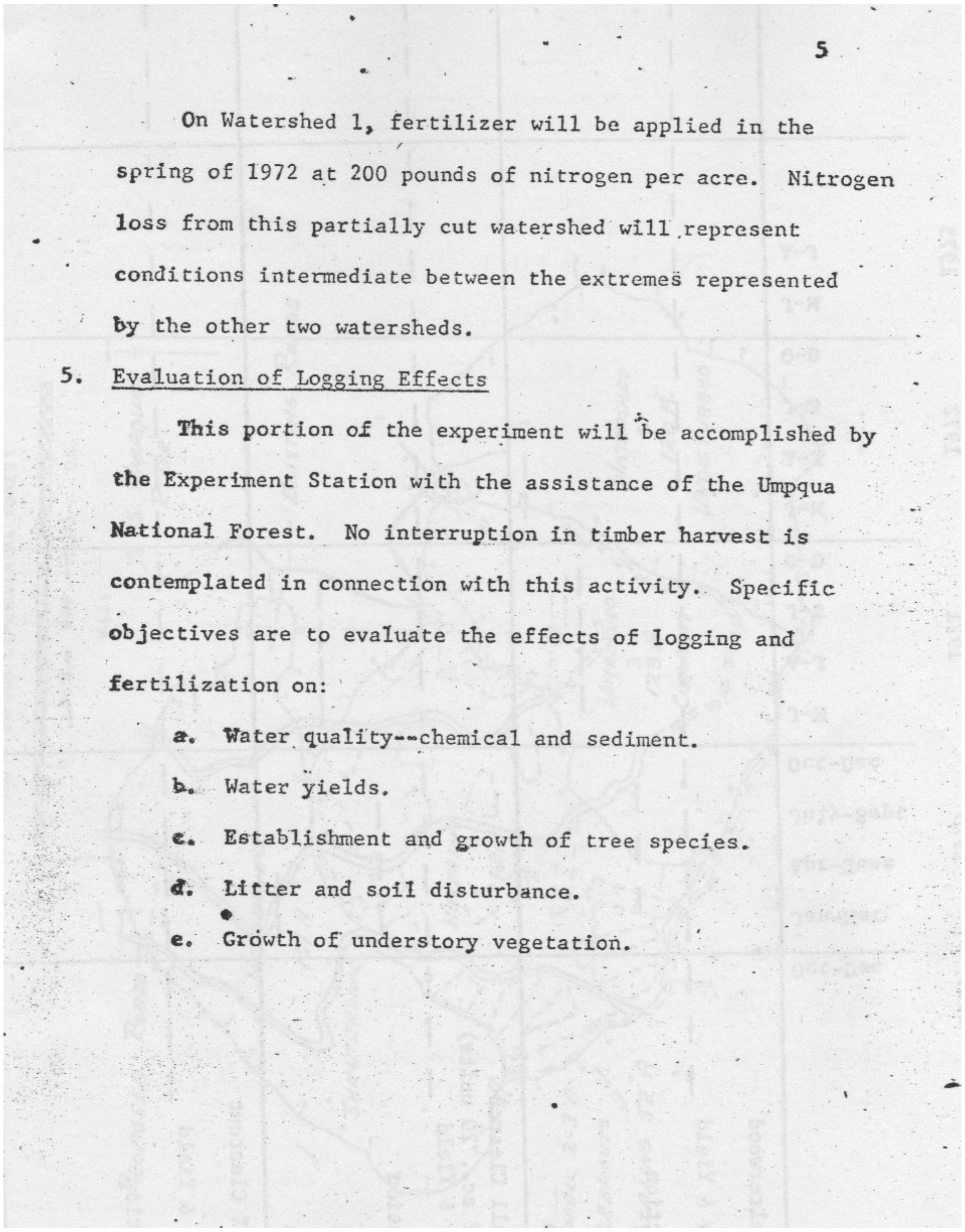
Watersheds 1 and 2 will be expected to regenerate naturally. If poor seed years are expected, units in Watershed 2 should be artificially seeded. Watershed 3 will be planted during the fall-spring seasons of 1971-72.

4. Fertilization

On Watershed 2, the Experiment Station will arrange with the Region and the Forest for application of 90kg 200 pounds of nitrogen per 0.4 ha acre to the entire watershed area in the spring of 1970--as soon after March 1 as possible. This application of fertilizer to an existing stand will represent the condition least likely to result in nitrate pollution of water.

On Watershed 3, fertilizer will be applied at the rate of 200 pounds of nitrogen per acre in the spring of 1973, a year and one-half after logging. Studies elsewhere indicate the possibility of nitrate pollution following cutting of timber even without fertilization. This will be studied from the fall of 1971 to spring of 1973. Fertilization after logging will represent the condition most likely to produce excessive concentrations of nitrogen in all forms in streamwater.

Fig. 6. Page 4 of study plan for fertilizer application and experiment (1969).



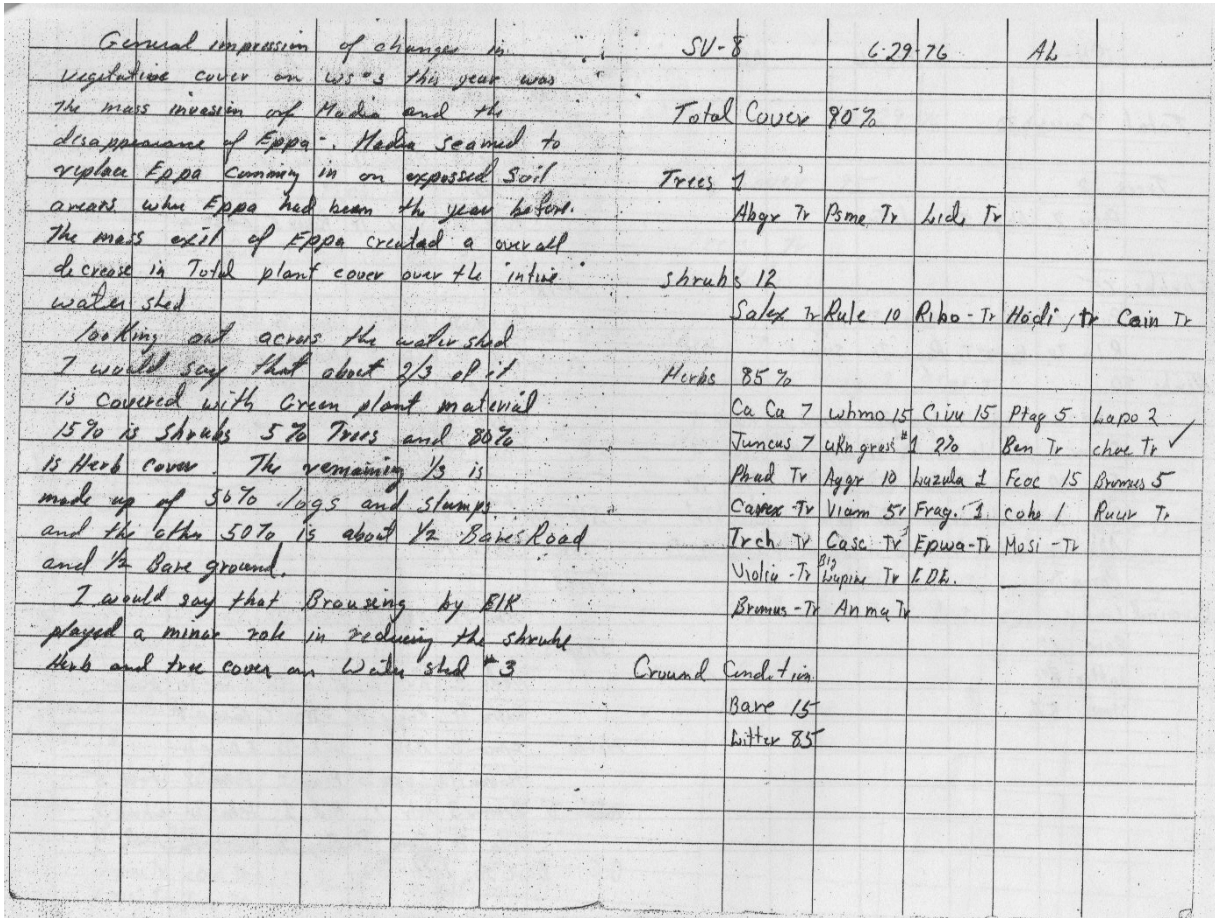
On Watershed 1, fertilizer will be applied in the spring of 1972 at 200 pounds of nitrogen per acre. Nitrogen loss from this partially cut watershed will represent conditions intermediate between the extremes represented by the other two watersheds.

5. Evaluation of Logging Effects

This portion of the experiment will be accomplished by the Experiment Station with the assistance of the Umpqua National Forest. No interruption in timber harvest is contemplated in connection with this activity. Specific objectives are to evaluate the effects of logging and fertilization on:

a. Water quality--chemical and sediment.
b. Water yields.
c. Establishment and growth of tree species.
d. Litter and soil disturbance.
e. Growth of understory vegetation.

Fig. 7. Page 5 of study plan for fertilizer application and experiment (1969).



General impression of change in vegetative cover on ws #3 this year was the mass invasion of Madia and the disappearance of Eppa. Madia seamed to replace Eppa comming in on exposed Soil areas when Eppa had been the year before. The mass exit of Eppa created a overall decrease in Total plant cover over the intire water shed

looking out across the water shed I would say that about 2/3 of it is covered with Green plant material 15% is Shrubs 5% Trees and 80% is Herb cover. The remaining 1/3 is made up of 50% logs and stumps and the other 50% is about 1/2 Bare Road and 1/2 Bare ground.

I would say that Brousing by Elk played a minor role in reducing the shrubs Herb and tree cover on Water shed #3

SV-8		6-29-76		AL	
Total Cover 90%					
Trees 1					
	Abgr Tr	Psme Tr	Lidi Tr		
Shrubs 12					
	Salex Tr	Rule 10	Ribo - Tr	Hodi 1 Tr	Cain Tr
Herbs 85%					
	Ca Ca 7	whmo 15	Civu 15	Ptag 5	Lapo 2
	Juncus 7	ukn grass #1	2%	Ben Tr	choe Tr ✓
	Phad Tr	Aggr 10	Luzula 1	Feoc 15	Bromus 5
	Carex - Tr	Viam 5	Frag. 1	coho 1	Ruur Tr
	Trch Tr	Casc Tr	Epwa - Tr	Mosi - Tr	
	Violia - Tr	Big Lupine Tr	E D L.		
	Bromus - Tr	Anma Tr			
Ground Condition					
	Bare 15				
	Litter 85				

Fig. 8. Sample of historical field book including survey format and notes (A. Levno 1976).

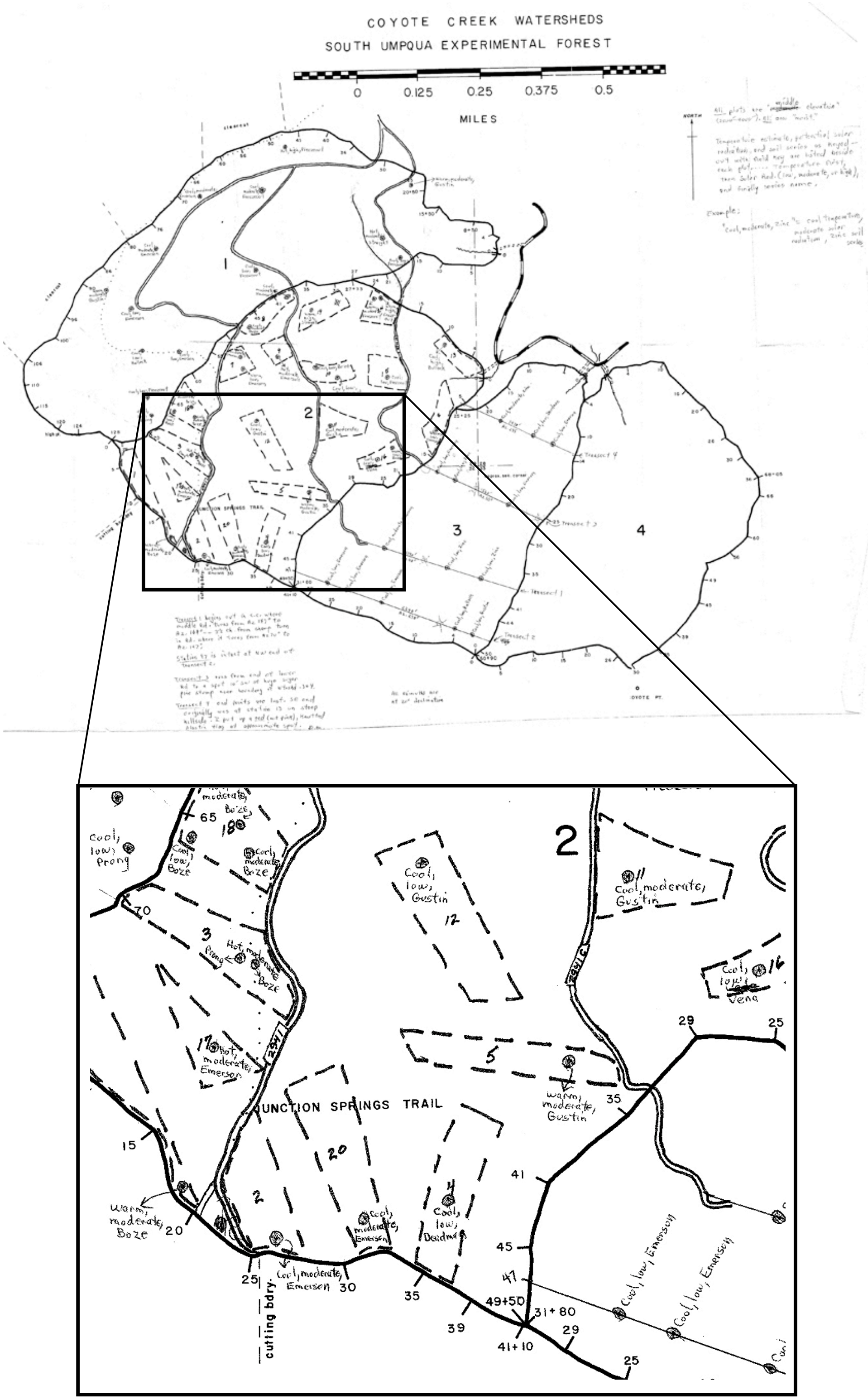


Fig. 9. Historical field map with close-up inset (c. 1971).

Appendix B. Species list

Table 1. Complete list of species surveyed in the South Umpqua Experimental Forest between 1971-1979, 1981, 2006 with common names, codes, succession and origin; nomenclature follows Hickman 1993.

Stratification	Species	Common name	Codes	Succession	Origin
Tree					
	Pseudotsuga menziesii (Mirb.) Franco	Douglas fir	PSME	residual	native
	Abies grandis (Douglas) Lindley	grand fir	ABGR	residual	native
	Calocedrus (*Libocedrus*) *decurrens* (Torrey) Florin	incense cedar	CADE/LIDE*	residual	native
	Tsuga heterophylla (Raf.) Sarg.	western hemlock	TSHE	residual	native
	Pinus lambertiana Douglas	sugar pine	PILA	residual	native
	Pinus ponderosa Laws.	ponderosa pine	PIPO	residual	native
	Arbutus menziesii Pursh	Pacific madrone	ARME	residual	native
Tall shrubs					
	Acer circinatum Pursh	vine maple	ACCI	residual	native
	Acer macrophyllum Pursh	big-leaf maple	ACMA	residual	native
	Amelanchier alnifolia (Nutt.) Nutt.	service berry	AMAL	residual	native
	Arctostaphylos canescens Eastw.	hoary manzanita	ARCA	invader	native
	Arctostaphylos columbiana Piper	hairy manzanita	ARCO	invader	native
	Castonopsis chrysophylla (Hook.) Hjelmq.	golden chinkapin	CACH^	residual	native
	Ceanothus integerrimus Hook. & Arn.	deer brush	CEIN	invader	native
	Ceanothus sanquineus Pursh	redstem	CESA	invader	native
	Ceanothus velutinus Hook.	tobacco brush	CEVE	invader	native
	Corylus cornuta Marsh	hazelnut	COCO	residual	native
	Cornus nuttallii Audubon	Pacific dogwood	CONU	residual	native
	Fraxinus latifolia Benth	Oregon ash	FRLA	invader	native
	Holodiscus discolor (Pursh) Maxim.	oceanspray	HODI	residual	native
	Prunus emarginata (Hook.) Walp.	bitter cherry	PREM	invader	native
	Ribes lobbii A. Gray	gummy gooseberry	RILO	invader	native
	Rhododendron macrophyllum D. Don	rhododendron	RHMA	residual	native
	Rhamnus purshiana DC.	cascara	RHPU	invader	native
	Rosa gymnocarpa Nutt.	baldhip rose	ROGY	residual	native
	Rosa nutkana Presl.	nootka rose	RONU	residual	native
	Rubus leucodermis Torrey & A. Gray	blackcap raspberry	RULE	invader	native
	Rubus parviflorus Nutt.	thimbleberry	RUPA	invader	native
	Sambucus racemosa L.	red elderberry	SARA	invader	native
	Salix scouleriana Hook.	Souleri's willow	SASC	invader	native
	Symphoricarpus albus (L.) S.F. Blake	snowberry	SYAL	invader	native
	Taxus brevifolia Nutt.	Pacific yew	TABR	residual	native
	Toxicodendron diversilobum (Torrey & A. Gray) E. Greene	poison oak	TODI	residual	native
	Vaccinium parvifolium Smith	red huckleberry	VAPA	residual	native
Short shrubs					
	UNKNOWN	UNKNOWN	BACH	invader	UNKNOWN
	Baccharis pilularis DC.	coyote brush	BAPI	invader	native
	Berberis aquifolium Pursh	tall Oregon grape	BEAQ	residual	native
	Berberis nervosa Pursh	dwarf Oregon grape	BENE	residual	native
	Gaultheria shallon Pursh	salal	GASH	residual	native
	Lonicera hispidula Douglas	honeysuckle	LOHI	residual	native
	Ribes sanguineum Pursh	red currant	RISA	invader	native
	Rubus discolor Weihe & Nees	Himalayan blackberry	RUDI	invader	exotic
	Rubus nivalis Douglas	snow bramble	RUNI	residual	native
	Rubus ursinus Cham. & Schldl.	trailing blackberry	RUUR	residual	native
	Symphoricarpos mollis Nutt.	creeping snowberry	SYMO	invader	native
Graminoids					
	Bromus orcuttianus (Shear) A. Hitchc.	Orcutt's brome	BROR	residual	native
	Calamagrostis canadensis (Michx.) Beauv.	reedgrass	CACA	invader	native
	Festuca occidentalis Hook.	western fescue	FEOC	residual	native
	Melica harfordii Bolander	oniongrass	MEHA	invader	native
	Phleum pratense L.	Timothy	PHPR	invader	exotic

Stratification	Species	Common name	Codes	Succession	Origin
	Poa spp	bluegrasses	*Poa*	invader	variable
	Achnatherum occidentalis (Stipa occidentalis) Thurber	needlegrass	STOC*	invader	native
	grass spp	various	grass	invader	various
	Carex spp	sedges	*Carex*	invader	variable
	Juncus spp	rushes	*Juncus*	invader	variable
	Luzula spp	wood rushes	*Luzula*	invader	native
Forbs					
	Achlys triphylla (Smith) DC.	vanilla leaf	ACTR	residual	native
	Actaea rubra (Aiton) Willd.	baneberry	ACRU	residual	native
	Adenocaulon bicolor Hook.	pathfinder	ADBI	residual	native
	Agoseris aurantica (Hook.) E. Greene	large flowered agoseris	AGAU	invader	native
	Agoseris grandiflora (Nutt.) E. Greene	orange agoseris	AGGR	invader	native
	Anaphalis margaritacea (L.) Benth. & Hook.	pearly everlasting	ANMA	invader	native
	Anemone deltoidea Hook.	anemone	ANDE	residual	native
	Arnica latifolia (gracilis) Bong.	mountain arnica	ARGR*	invader	native
	UNKNOWN	UNKNOWN	ARHA	invader	UNKNOWN
	Asarum caudatum Lindley	wild ginger	ASCA	residual	native
	Asarum hartwegii S. Watson	Harweg's ginger	ASHA	residual	native
	Athyrium filix-feminia (L.) Roth	lady fern	ATFI	residual	native
	UNKNOWN	UNKNOWN	BAHE	invader	UNKNOWN
	Blechnum spicant (L.) Smith	deer fern	BLSP	residual	native
	Boykinia elata (Nutt.) E. Greene	boykinia	BOEL	residual	native
	Calochortus spp	mariposa lilies	*Calo.*	invader	native
	Calypso bulbosa (L.) Oakes	fairy slipper	CABU	residual	native
	Campanula prenanthoides Durand	harebell	CAPR	invader	native
	Campanula scouleri Engelm.	Scouler's harebell	CASC	residual	native
	Cardamine tenella (Pursh) O. Schulz	bittercress	CATE	invader	exotic
	Centaurium erythraea Raf.	centaury	CEER	invader	exotic
	Chimaphila menziesii D. Don	little prince's pine	CHME	residual	native
	Chimaphila umbellatum (L.) Bartram	prince's pine	CHUM	residual	native
	Chrysanthemum leucanthemum L.	ox-eye daisy	CHLE	invader	exotic
	Cirsium vulgare (Savi) Ten.	bull thistle	CIVU	invader	exotic
	Claytonia perfoliata Willd.	miner's lettuce	CLPE	invader	native
	Claytonia sibirica L.	candyflower	CLSI	invader	native
	Clintonia uniflora (Scultes) Kunth	queen's cupt	CLUN	residual	native
	UNKNOWN	UNKNOWN	COAG	invader	UNKNOWN
	Collomia heterophylla Hook.	collomia	COHE	invader	native
	Conyza canadensis (L.) Cronq.	horseweed	COCA	invader	native
	Corallorhiza maculata Raf.	spotted coralroot	COMA	residual	native
	Cynoglossum grande Lehm.	hound's tongue	CYGR	invader	native
	Daucus carota L.	Queen Anne's lace	DACA	invader	exotic
	Disporum hookerai (Torrey) Nicholson	fairy bells	DIHO	residual	native
	Disporum smithii (Hook.) Piper	fairy bells	DISM	residual	native
	Epilobium angustifolium L.	fireweed	EPAN	invader	native
	Epilobium brachycarpum C. Presl	willow-weed	EPBR	invader	native
	Epilobium minutum Lehm.	small flower willow-weed	EPMI	invader	native
	Epilobium paniculatum Torrey & A. Gray	willow-weed	EPPA	invader	native
	Epilobium watsonii Barbey	Watson's willow-weed	EPWA	invader	native
	Equisetum telmateia Ehrh.	tall horsetail	EQTE	residual	native
	Fragaria vesca (californica) L.	wild strawberry	FRCA	invader	native
	Fragaria virginiana Duchesne	wild strawberry	FRVI	invader	native
	Gallium oreganum Britton	bedstraw	GAOR	residual	native
	Gallium triflorum Michaux	bedstraw	GATR	residual	native
	Geranium molle L.	dove's foot geranium	GEMO	invader	exotic
	Geranium spp	geraniums	*Gera.*	invader	variable
	Gnaphalium microcephalum Nutt.	cudweed	GNMI^	invader	native
	Goodyera oblongifolia Raf.	rattlesnake plantain	GOOB	residual	native
	Hieracium albiflorum Hook.	hawkweed	HIAL	residual	native
	Hypericum perfoliatum L.	Klamathweed	HYPE	invader	exotic
	Hypochaeris radicata L.	rough cat's ear	HYRA	invader	exotic
	Iris chrysophylla Howell	yellow iris	IRCH	invader	native
	Iris tenax Douglas	tough leaved iris	IRTE	invader	native

Stratification	Species	Common name	Codes	Succession	Origin
	UNKNOWN	UNKNOWN	LACE	invader	UNKNOWN
	Lactuca serriola L.	prickly lettuce	LASE	invader	exotic
	Lathyrus palustris L.	marsh pea	LAPA	invader	native
	Lathyrus polyphyllus Nutt.	wild pea	LAPO	invader	native
	Ligusticum apiifolium (Nutt.) A. Gray	licource root	LIAP	invader	native
	Ligusticum grayi J. Coulter & Rose	licource root	LIGR	invader	native
	Lillium columbianum Baker	Columbian lily	LICO	invader	native
	Lillium washingtonianum Kellogg	Shasta lily	LIWA	invader	native
	Linnea borealis L.	twinflower	LIBO	residual	native
	Linum spp	flaxes	*Linum*	invader	variable
	Listera cordata (L.) R.Br.	twayblade	LICO1	invader	native
	Lotus crassifolius (Benth.) E. Greene	lotus	LOCR	invader	native
	Lupinus spp	lupines	*Lupinus*	invader	variable
	Madia madiodes (Nutt.) E. Greene	tarweed	MAMA	invader	native
	Mertensia spp	bluebells	*Mert.*	invader	native
	Mimulus moschatus Lindley	must monkeyflower	MIMO	residual	native
	UNKNOWN	UNKNOWN	MOTA	invader	UNKNOWN
	Nemophila parviflora Benth.	small flowered nemophila	NEPA	residual	native
	Osmorhiza chilensis Hook. & Arn.	sweet cicely	OSCH	residual	native
	Phlox adsurgens A. Gray	plox	PHAD	residual	native
	Pentagramma (Pityrogramma) triangularis (Kaulf.) Maxon	silver fern	PITR	residual	native
	Plantago major L.	common plantain	PLMA	invader	exotic
	Polystichum munitum (Kaulf.) C. Presl	swordfern	POMU	residual	native
	Prunella vulgaris L.	self-heal	PRVU	invader	exotic
	Pteridium aquilinium (L.) Kuhn	bracken fern	PTAQ	invader	native
	Pyrola asarifolia Michaux	wintergreen	PYAS	residual	native
	Pyrola chlorantha Sw.	green wintergreen	PYCH	residual	native
	Pyrola picta Smith	white wintergreen	PYPI	residual	native
	Ranunculus uncinatus D. Don	woodland buttercup	RAUN	residual	native
	Rumex acetosella L.	sheep sorrel	RUAC	invader	exotic
	Rupertia physodes (Hook.) Grimes	California tea	RUPH	invader	native
	Satureja douglasii (Benth.) Briq.	yerba buena	SADO	residual	native
	Senecio sylvaticus L.	ragwort	SESY	invader	exotic
	Smilacina stellata (L.) Desf.	false Solomon's seal	SMST	residual	native
	Stachys cooleyae Heller	hedge nettle	STCO^	residual	native
	Synthyris reniformis (Douglas) Benth.	snow queen	SYRE	residual	native
	Thermopsis gracilis Howell	golden pea	THGR	invader	native
	Tiarella trifoliata L.	lace flower	TITR	residual	native
	Trientalis latifolia Hook.	starflower	TRLA	residual	native
	Trifollium dubium Sibth.	suckling clover	TRDU^	invader	exotic
	Trillium ovatum Pursh	western trillium	TROV	residual	native
	Vancouveria hexandra (Hook.) Morren & Decne.	inside-out flower	VAHE	residual	native
	Vicia americana Willd.	American vetch	VIAM	invader	native
	Viola glabella Nutt.	wood violet	VIGL	residual	native
	Viola sempervirens E. Greene	evergreen violet	VISE	residual	native
	Whipplea modesta Torrey	whipplea	WHMO	residual	native
	Xerephyllum tenax (Pursh) Nutt.	beargrass	XETE	residual	native

* = Field codes differ from nomenclature herein
^ = follows Hitchcock & Cronquist

Appendix C. Plot Characteristics

Table 2. Summary of plot characteristics (all years surveyed = 1971-1979,1981, 2006; italics = estimated).

WS No.	Plot No.	Aspect degrees	Slope (%)	Elev. (m)	Soil disturbance	Vegetation community	Years surveyed
1	21	*3*	*35*	*948*	N/A	PSME-ABGR/GASH	1971
1	22	*20*	*25*	*949*	N/A	PSME-ABGR/GASH	1971
1	23	*100*	*35*	*963*	N/A	PSME/BROR-WHMO	1971
1	25	*55*	*40*	*835*	N/A	PSME-TSHE/GASH	1971
1	36	*15*	*60*	*908*	N/A	PSME-ABGR/GASH	1971
1	37	*110*	*60*	*902*	N/A	PSME-ABGR/GASH	1971
1	38	*135*	*55*	*930*	N/A	PSME-ABGR/GASH	1971
1	39	*95*	*35*	*943*	N/A	PSME/BROR-WHMO	1971
1	40	*160*	*70*	*920*	N/A	PSME/BROR-WHMO	1971
1	49	*110*	*45*	*829*	N/A	PSME/BROR-WHMO	1971
1	50	*150*	*50*	*823*	N/A	PSME/BROR-WHMO	1971
1	71	76	20	825	N/A	N/A	2006
1	72	114	15	854	N/A	N/A	2006
1	73	144	25	866	N/A	N/A	2006
1	74	148	25	911	N/A	N/A	2006
1	75	32	35	907	N/A	N/A	2006
1	76	118	15	896	N/A	N/A	2006
1	77	165	15	907	N/A	N/A	2006
1	78	165	35	940	N/A	N/A	2006
1	79	355	25	922	N/A	N/A	2006
1	80	105	15	934	N/A	N/A	2006
1	81	129	10	930	N/A	N/A	2006
1	82	347	10	940	N/A	N/A	2006
1	83	150	10	936	N/A	N/A	2006
1	84	140	10	967	N/A	N/A	2006
1	85	315	15	993	N/A	N/A	2006
1	86	20	15	966	N/A	N/A	2006
1	87	355	15	982	N/A	N/A	2006
1	88	26	20	994	N/A	N/A	2006
2	1	34	15	940	mod	PSME/BROR-WHMO	all
2	2	115	25	951	light	PSME/BROR-WHMO	all
2	3	80	20	954	light	PSME-ABGR/HODI-GASH	all
2	4	136	50	989	N/A	PSME/BROR-WHMO	1971/2006
2	5	110	10	979	heavy	PSME/BROR-WHMO	all
2	6	152	10	976	light	unclassified	all
2	7	58	10	966	heavy	PSME-ABGR/GASH	all
2	8	86	15	957	mod	PSME-ABGR/GASH	all
2	9	46	25	960	heavy	PSME-ABGR/GASH	all
2	10	93	20	923	heavy	PSME-ABGR/GASH	all
2	11	88	25	968	light	PSME/BROR-WHMO	all

WS No.	Plot No.	Aspect degrees	Slope (%)	Elev. (m)	Soil disturbance	Vegetation community	Years surveyed
2	12	56	20	973	mod	PSME/BROR-WHMO	all
2	13	25	30	927	heavy	PSME-ABGR/HODI-GASH	all
2	14	88	20	889	mod	PSME-ABGR/HODI-GASH	all
2	15	11	55	889	light	PSME-TSHE/GASH	all
2	16	93	*25*	916	mod	PSME-ABGR/GASH	all
2	17	121	*45*	890	light	PSME/BROR-WHMO	all
2	18	135	25	867	mod	PSME-ABGR/HODI-GASH	all
2	19	98	15	866	mod	PSME-ABGR/GASH	all
2	20	58	40	888	light	PSME-ABGR/GASH	all
2	23A	*135*	*30*	*780*	N/A	PSME-ABGR/HODI-GASH	1974-1981
3	24	289	30	811	light	PSME-TSHE/GASH	all
3	26	36	40	870	mod	PSME-TSHE/GASH	all
3	27	82	10	882	heavy	PSME-TSHE/GASH	all
3	28	5	15	854	light	PSME-TSHE/GASH	all
3	29	330	*15*	872	light	PSME-ABGR/HODI-GASH	all
3	30	22	35	914	mod	unclassified	all
3	31	360	35	900	mod	unclassified	all
3	32	25	5	935	heavy	unclassified	all
3	33	109	55	954	light	unclassified	all
3	34	34	50	941	light	PSME-ABGR/GASH	all
3	35	90	25	945	light	unclassified	all
3	41	50	15	834	mod	PSME-TSHE/GASH	all
3	42	348	45	823	light	unclassified	all
3	43	27	50	837	heavy	PSME-ABGR/HODI-GASH	all
3	44	301	40	821	N/A	PSME-TSHE/GASH	1971/2006
3	45	327	10	862	N/A	unclassified	all (-1972)
3	46	339	10	761	heavy	PSME-TSHE/GASH	all
3	47	51	30	776	heavy	PSME-TSHE/GASH	all
3	48	116	15	775	mod	unclassified	all
4	51	75	20	863	N/A	N/A	2006
4	52	117	30	843	N/A	N/A	2006
4	53	43	25	812	N/A	N/A	2006
4	54	90	35	834	N/A	N/A	2006
4	55	42	50	850	N/A	N/A	2006
4	56	218	5	844	N/A	N/A	2006
4	57	14	50	886	N/A	N/A	2006
4	58	3	30	862	N/A	N/A	2006
4	59	335	90	862	N/A	N/A	2006
4	60	278	40	862	N/A	N/A	2006
4	61	212	40	887	N/A	N/A	2006
4	62	89	10	852	N/A	N/A	2006
4	63	82	40	842	N/A	N/A	2006
4	64	311	25	803	N/A	N/A	2006
4	65	281	35	800	N/A	N/A	2006
4	66	287	45	839	N/A	N/A	2006

WS No.	Plot No.	Aspect degrees	Slope (%)	Elev. (m)	Soil disturbance	Vegetation community	Years surveyed
4	67	105	10	786	N/A	N/A	2006
4	68	282	10	766	N/A	N/A	2006
4	69	233	20	783	N/A	N/A	2006
4	70	250	45	811	N/A	N/A	2006

Appendix D. Memorandum from Don Goheen to the Umpqua National Forest

USDA Forest Service
Southwest Oregon Forest Insect and Disease Service Center
2606 Old Stage Road
Central Point, OR
97502

File Code: 3400

Date: January 2, 2001 [sic]

To Kathy Minor
Tiller Ranger District
Umpqua National Forest

On September 6, 2001 [sic], I visited the South Umpqua Experimental Forest for the purpose of evaluating the condition of pines. This was a one-day reconnaissance involving a general drive-through and walk-through examination of stands in the area and was not a formal survey. Findings were as follows:

1) Ponderosa pines (*Pinus ponderosa*) are significant components of stands in the Experimental Forest. They are common in scattered locations in the eastern part of the area, especially on the upper slope and ridge top above road 600. Ponderosa pines in unentered and partially cut stands in this area are large, many with diameters between 36 and 48 inches at breast height. Large ponderosa pines also occur but in smaller numbers and at much more widely scattered locations in the central and western portions of the Forest as well. Young ponderosa pines are numerous along most roads in the Experimental Forest and in many of the plantations developed in the old patch cuts done in the south central portion of the Forest. Many of the plantation pines were obviously planted because most of the plantations are situated at considerable distances from natural ponderosa pine seed sources.
2) Sugar pines (*Pinus lambertiana*) are even more important stand components than ponderosa pines in the mature stands in the experimental forest. Large trees are common to very common throughout the Forest. Numbers are highest on the ridge tops and upper slopes, but sugar pines are present in some numbers down all the way into the riparian areas along Coyote Creek. Sugar pines in unentered or partially cut stands are very large, many with diameters between 50 and 70 inches at breast height. I examined one intermediate-age thinned stand that contained some sugar pines at the end of Road 600.

Sugar pines in this stand were 10 to 20 inches dbh. I observed young sugar pine regeneration at scattered locations along roads and in understories in old partial cuts. However, sugar pines are very rare to nonexistent in the young plantations in the patch cuts.

3) There is considerable mortality of sugar pines over much of the Experimental Forest. I observed current sugar pine mortality as well as many sugar pines that have died over the last decade. Death of large sugar pines appears to be mostly due to infestation by mountain pine beetle (*Dendroctonus ponderosae*) while seedlings and saplings have been killed primarily by white pine blister rust (caused by the fungus *Cronartium ribicola*).
4) In the absence of appropriate management activities, the prognosis for the long-term health of both sugar and ponderosa pines in the Experimental Forest is not good. Sugar pines in particular appear to be at imminent risk. Dangerously high stand densities and high hazard for blister rust are major concerns.
5) Pine bark beetles (*D. ponderosae* and *D. brevicomis*) preferentially infest medium-sized to large pines (10 inch dbh or greater) that are under a certain amount of stress. Fire injured, diseased, or lightning struck pines are preferred hosts as are pines that are growing in overly dense stands where competition with other trees reduces their vigor. Overstocking is considered the major factor that predisposes pines to beetle attack in Southwest Oregon. Prior to European settlement, repeated light natural or native American-caused fires regulated stand densities to some degree and minimized high levels of pine bark beetle activity in many areas. The last 40 to 60 years of fire exclusion have allowed substantially higher stand densities to develop, resulting in much greater bark beetle risk to the remaining pines. The threshold stand density beyond which the pine components of a stand become vulnerable to bark beetle infestation differs with site quality (that is pines on good sites can tolerate somewhat greater densities than those on poor sites). On sites such as those at the South Umpqua Experimental Forest, I would suggest that the likelihood of beetle infestation becomes significant when basal areas around pines exceed 150 square feet per acre. Current densities in the Experimental Forest greatly exceed that threshold. Basal areas around a randomly selected sample of ponderosa pines (12 trees) averaged 180 square feet per acre (range 160 to 200 square feet per acre) while basal areas around a sample of sugar pines (24 trees) averaged 320 square feet per acre (range 240 to 360 square feet per acre). Furthermore, the large numbers of small trees (less than 5 inches dbh) around virtually all pines examined indicates that densities will be very much higher in the future. If stand densities are not reduced around pines in the Experimental Forest, bark beetle-caused mortality will

continue in the sugar pine and almost certainly will also become important in the ponderosa pine.

6) White pine blister rust is a virulent, exotic disease that was introduced into western North America in 1910. It affects all five-needle pines and has spread rapidly through the ranges of its hosts. From an initial introduction point in Vancouver, British Columbia, it reached Southwest Oregon in the late 1920s. On five-needle pines, *C. ribicola* causes formation of resinous lesions that can girdle the host at the point of infection. This results in branch and top mortality of large trees, and outright death of small trees when infections are on the main stems. *C. ribicola* has a complex life cycle involving five spore stages and requiring two hosts, a five-needle pine, and a shrub in the genus *Ribes*. Windborne spores from infected pines infect *Ribes* leaves in spring, and spores from *Ribes* infect pine needles in late summer and early fall. Infection on both hosts is greatly favored by cool, moist conditions at the time of spore dispersal. This is especially true when spores are spreading from *Ribes* to pines in which case 48 hours with 100% relative humidity and temperatures not exceeding 68 degrees F are actual requirements for infection. Currently, the major control for white pine blister rust involves planting pines with proven genetic resistance to *C. ribicola*, especially on sites where *Ribes* populations exist and environmental conditions are conducive for infection. Pruning to reduce potential infection sites on lower boles is also often recommended in conjunction with planting resistant stock. The Dorena Genetic Resource Center at Cottage Grove can supply sugar pine planting stock that has various degrees of resistance to *C. ribicola* and at the same time is adapted for growth on most Southwest Oregon sites. Based on the currently observed condition of young sugar pines in the South Umpqua Experimental Forest, it appears that there are many high-risk sites there. In the absence of special efforts to establish resistant trees in the area, very few young sugar pines can be expected to survive to maturity.

7) If maintaining the health of pine stand components on the South Umpqua Experimental Forest is an important management object, I recommend that managers should:
    a) Consider thinning selected stands so that basal areas around pines are reduced to below 150 square feet per acre. Select healthy-appearing ponderosa and sugar pines with live crown ratios of 25% or greater as preferred leave trees. Exercise caution in location and design of thinning projects because onsite evidence suggests that the Douglas-fir and true fir components of stands in many areas on the experimental forest are very prone to windthrow. Monitor treated stands and use prescribed burning or

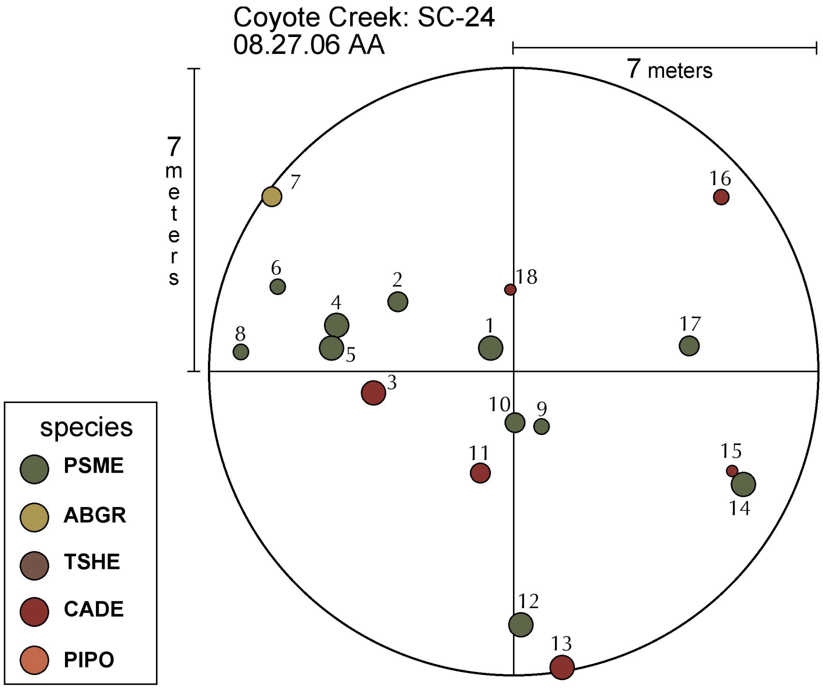
additional thinning treatments in the future to maintain stocking below the bark beetle high-risk threshold.

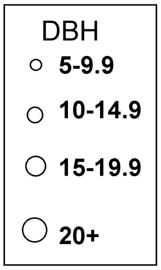
  - b) In areas where thinning of entire stands is not deemed appropriate, consider individual tree thinning around selected specimen ponderosa and sugar pines that are desired for long-term retention. Remove trees (including small ones) from below the crown and out to 25 feet beyond the drip line of the leave trees.
  - c) Consider planting blister rust resistant sugar pines in mixes with other species in any regeneration opportunities on the experimental forest. Once established, promote continued health of resistant trees through pruning and density management.

8) Opportunities to promote and maintain pine health are exceptionally great in this area. The Southwest Oregon Forest Insect and Disease Service Center would be delighted to assist the Tiller District in further evaluating stand conditions, considering treatment options, and designing monitoring strategies for stands in the South Umpqua Experimental Forest. Potential funding to facilitate analysis and treatments in the Experimental Forest may be available through Forest Health Protection Insect and Disease Suppression/Prevention Funds. Do not hesitate to call upon us for additional assistance on this or other forest insect and disease related questions.

/s/ Donald J. Goheen
Plant Pathologist/Entomologist
Southwest Oregon Forest Insect and Disease Service Center

Appendix E. Stem Count Sample





Tree #	Species	DBH (cm)	Azimuth	Distance (m)
1	PSME	26	317	0.58
2	PSME	17.8	301	2.95
3	CADE	22.5	262	3.08
4	PSME	20.7	284	3.99
5	PSME	20.6	279	3.99
6	PSME	10.2	290	5.61
7	ABGR	15.5	306	6.89
8	PSME	11.9	274	6.1
9	PSME	13.5	154	1.25
10	PSME	18.8	178	0.94
11	CADE	16.9	198	2.21
12	PSME	25.4	179	5.58
13	CADE	22.9	171	6.77
14	PSME	23.9	116	5.59
15	CADE	9.1	115	5.49
16	CADE	13.7	50	6.1
17	PSME	18.9	84	3.84
18	CADE	9.1	358	1.78

Fig. 10. Sample stem count map from Plot 24 in WS-2.