

# Structure and yield of two-aged stands on the Willamette National Forest, Oregon: implications for green tree retention

S.A. Acker, E.K. Zenner, and W.H. Emmingham

**Abstract:** Green tree retention, a practice based on ecosystem science, has been integrated into forest management as a working hypothesis, requiring research and monitoring to quantify its effects. We undertook a retrospective study of natural, two-aged forest stands on the Willamette National Forest to provide preliminary estimates of the effects of green tree retention on forest structure and yield. Fourteen stands in the western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) zone (mostly dominated by Douglas-fir, *Pseudotsuga menziesii* (Mirb.) Franco) were sampled using adjacent, paired plots: one with residual trees over a younger cohort (median age of younger cohort, 97 years) and one with only the younger cohort. The younger cohorts all had much higher densities of trees than typical of local Douglas-fir plantations. Basal area, volume, and mean annual increment (MAI) of the younger cohort all declined with increasing residual tree basal area. These relationships were best described by curvilinear models; the greatest effect per unit of residual tree basal area occurred at low residual tree levels (5–10 m<sup>2</sup>/ha). For 10 m<sup>2</sup>/ha residual tree basal area (equivalent to about 12 average (diameter at breast height 105 cm) residual trees per hectare), we predicted a 26% decline in younger cohort MAI (95% confidence interval: –30% to –22%).

**Résumé :** La technique sylvicole de réserves-sur-coupe, une pratique fondée sur la science de l'écosystème, a été intégrée à l'aménagement comme hypothèse de travail sans qu'aucune vérification n'ait permis de quantifier ses effets. Nous avons entrepris une étude rétrospective des peuplements naturels bi-étagés issus de cette technique sylvicole dans la Forêt nationale de Willamette pour évaluer de façon préliminaire ses effets sur la structure et le rendement. Des placettes adjacentes et paires ont été installées dans 14 peuplements faisant partie du domaine de la pruche de l'ouest (*Tsuga heterophylla* (Raf.) Sarg.) dominé par le Douglas taxifolié (*Pseudotsuga menziesii* (Mirb.) Franco) : une placette dans le peuplement bi-étagé et une autre dans le peuplement constitué uniquement du jeune étage dont l'âge médian était de 97 ans. Les jeunes étages étaient beaucoup plus denses que les plantations typiques de *Pseudotsuga menziesii*. La surface terrière, le volume et l'accroissement annuel moyen du jeune étage ont tous diminué avec l'augmentation de la surface terrière de la réserve. Ces relations étaient mieux décrites par les modèles non linéaires. L'effet le plus marqué par unité de surface terrière de la réserve a été observé dans les faibles réserves (5–10 m<sup>2</sup>/ha). Lorsque la réserve atteignait 10 m<sup>2</sup>/ha de surface terrière (équivalente à environ 12 réserves de taille moyenne à l'hectare (105 cm de diamètre à hauteur de poitrine), on pouvait prédire une diminution de 26% en accroissement annuel moyen du jeune étage (intervalle de confiance à 95% compris entre –30% et –22%).

[Traduit par la Rédaction]

## Introduction

Integration of ecosystem science into policy and practice is one of the fundamental changes occurring in management of federal forests in the United States (Kessler et al. 1992). That biotic structures may play critical roles in persistence of ecosystems through catastrophic disturbances is one result from ecosystem science that has stimulated changes in management (Franklin 1988, 1989; Swanson and Franklin 1992). Among such "legacies" are snags, logs, patterns in soil chemistry, plant propagules, and live trees (Swanson and Franklin 1992). Altering timber harvest to leave some large, live trees has been advocated as an application of the legacy concept (e.g., Franklin 1989). "Green tree retention," as this practice is called, has

recently been adopted as a standard policy for federal forests within the range of the northern spotted owl (*Strix occidentalis caurina*; USDA Forest Service and USDI Bureau of Land Management 1994) and has also been mandated for state and private timberlands in Oregon (State of Oregon Forest Practices Act, 1991 revision).

Although green tree retention and other ideas from ecosystem science are being incorporated into forest management, the implications of these changes are uncertain (Swanson and Franklin 1992). Rather than waiting for decades of ecosystem development under new management practices, it is critical to marshal as much information as possible now on the potential implications of green tree retention (Thomas et al. 1993). To this end, we conducted a retrospective study of patchy stands that developed after natural disturbances as analogues for green tree retention harvest.

In comparison to manipulative experiments, retrospective studies provide information more quickly but may be less definitive. Interpretation of retrospective data requires assumptions concerning the history of study sites, assumptions that may be difficult or impossible to test. However, in the case of forest management, retrospective studies can produce information

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decades before the conclusion of experiments. Thus, retrospective studies can serve as a means to gain understanding in the short term (Foster et al. 1996) and are a useful complement to long-term experiments.

Drawing inferences for managed stands from any study of natural stands requires caution. Conditions during stand development may be different because of artificial regeneration of trees, removal of competing vegetation, and control of tree density. On the other hand, understanding natural ecosystems may be an important prerequisite to sustainable land management (Lubchenco et al. 1991; Swanson and Franklin 1992). Furthermore, natural processes may dominate some managed forests in the future if land managers seek to minimize inputs (Swanson and Franklin 1992).

In addition to providing information on natural stands that may serve as models for stands managed with green tree retention, our objective was to test the effects of residual trees on stand development. We tested the effect of the residual trees on standing basal area, tree density, standing volume, mean annual increment (MAI), and average height of the younger cohort of trees beneath the older component of the stands. We also tested the effect of the older component on standing live volume of the younger cohort and older component combined. Other aspects of the vegetation of these stands have been analyzed elsewhere (i.e., understory plants, Traut 1994; epiphytes, Peck and McCune 1997). We have also analyzed data on the tree stratum with an alternative set of assumptions and techniques (Zenner 1995; Zenner et al. 1998).

As green tree retention has been implemented, its operational definition has changed. In the late 1980s and early 1990s, green tree retention was described in terms of retained trees per acre (1 acre = 0.405 ha; e.g., Franklin 1989; Birch and Johnson 1992), often with trees dispersed throughout cutting units (e.g., Franklin and Spies 1991). Currently, however, prescriptions for federal timberlands in western Oregon and Washington contain at least two distinct methods of green tree retention (USDA Forest Service and USDI Bureau of Land Management 1994). On Bureau of Land Management lands in Oregon, green trees are "scattered" throughout the cutting unit. For most National Forests in western Oregon and Washington, green trees will be aggregated in patches 0.2 ha or larger. This study concerns natural, two-aged stands with scattered residual trees and so is not directly applicable to aggregated green tree retention.

## Methods

### Site selection and study areas

We studied 14 pairs of plots in two-aged stands within the western hemlock zone on the Willamette National Forest, spanning most of the north-south extent of the forest (from 43°48' to 44°43'N). Study sites were selected after on-site inspection of an extensive list of potential study areas throughout the forest. We sought areas of unmanaged forests 60–100 years old (referred to below as "younger cohort"), with scattered large trees that survived the stand-initiating disturbance ("residual trees"). Our sampling design required areas of at least 1 ha that were relatively homogeneous (i.e., no major slope breaks, drainages, obvious soil differences, or patches of tree mortality). Sites were included in the study when it was possible to locate two, 61.8 m diameter circular plots entirely on one topographic feature, one plot with large residual trees and the other without.

Plots represented a narrow range of environments as indicated by

plant associations (Hemstrom et al. 1987) (Table 1). All but one of the pairs of plots belonged to the western hemlock (*Tsuga heterophylla* (Raf.) Sarg./dwarf Oregon grape (*Berberis nervosa* Pursh) or closely related western hemlock/dwarf Oregon grape – salal (*Gaultheria shallon* Pursh) plant associations. These two plant associations are very common on the Willamette National Forest and blend into one another along a narrow moisture gradient (Hemstrom et al. 1987).

As indicated by site index of the younger cohort, plots included a range of moderate to low productivity levels (Table 1). Site index ranged from 19.5 to 34.7 m at 50 years (King 1966). Only three of the pairs of plots had values as high as the averages for their plant association in Hemstrom et al. (1987). Paired plots represented site class 3 (i.e., medium productivity; seven pairs), site class 4 (i.e., moderately low productivity; six pairs), and site class 5 (i.e., low productivity; one pair) (King 1966).

Density of live residual trees covered a broad range, from 3 to 57/ha, a somewhat broader range than green tree retention prescriptions in recent National Forest plans for western Oregon (Birch and Johnson 1992). However, for the majority of the plots, live residual tree densities fell within the range of the green tree retention prescriptions (i.e., 5–25 trees/ha) (Table 2).

Younger cohort stand ages ranged from 72 to 139 years and were similar within pairs (Table 1). Within pairs, plots had the same elevation and topographic position and were quite similar in slope and aspect (Table 1). There were no consistent patterns of difference between paired plots in either litter depth or surface soil pH (data not shown). Mean litter depth was greater on plots with residual trees for 9 of the 14 pairs and greater on plots without residual trees for the other 5 pairs. Mean surface soil pH was higher on plots with residual trees for 7 of the 14 pairs of plots and higher on plots without residual trees for the other 7 pairs.

### Sampling design

We used nested, concentric, fixed-radius plots to measure residual trees and the younger cohort. Residual trees were measured in the larger of the plots ("residual plots") and the younger cohort in the smaller of the plots ("detection plots"). Nested plots were used to characterize the large residual trees that affected the development of the younger cohort. Although trees may affect one another over large distances, we assumed that the effects of a residual tree on trees in the younger cohort were minimal at and beyond 18.29 m (Hoyer 1993). We used detection plots of 500 m<sup>2</sup> (radius 12.62 m), and residual plots of 3000 m<sup>2</sup> (radius 30.90 m). Plot areas were slope corrected in the field. Paired residual plots (one with residual trees, one without) were placed as close together as possible.

### Data collection

For each residual plot, various environmental data were recorded in the field or obtained from maps. Slope, aspect, and topographic position (bottom, middle, or top one-third of slope) were recorded in the field. Litter depth and surface soil pH were measured as part of a concurrent study of understory vegetation (Traut 1994). Plant association (Hemstrom et al. 1987) was determined from understory plant data (Traut 1994). Elevation was determined from topographic maps.

We recorded species and diameter at breast height (DBH; breast height = 1.37 m) for all trees  $\geq$  5 cm DBH, living or dead, residual or younger cohort. Additional measurements depended on whether the tree was alive or dead and on whether the tree was a residual or of the younger cohort. All residual trees within the residual plot were measured; younger cohort trees were measured within the detection plot only. For residuals we obtained four perpendicular crown radii. We measured total height and height to base of live crown for all residuals and a representative sample of the younger cohort. We extracted increment cores from a representative sample of the younger cohort to determine the age of the younger cohort. We also extracted increment cores of at least 120 years from all residual trees to estimate basal area

**Table 1.** Environmental and ecological characteristics of plots.

Plot	W/O <sup>a</sup>	Elevation (m)	Aspect (°)	Slope (%)	Topography <sup>b</sup>	Plant association <sup>c</sup>	Site index <sup>d</sup>	Residual trees/ha <sup>e</sup>	Stand age <sup>f</sup>	Relative density <sup>g</sup>	Log volume <sup>h</sup>
BR07	W	823	14	85	B	TSHE/BENE	23.2	33	126	0.38	303
BR07	O	823	29	60	B	TSHE/BENE	23.2		139	0.98	132
BR08	W	823	350	75	B	TSHE/BENE – TSHE/RHMA–BENE	19.5	57	133	0.53	66
BR08	O	823	343	80	B	TSHE/BENE – TSHE/RHMA–BENE	19.5		132	0.77	109
BR09	W	671	210	75	T	TSHE/BENE	29.6	13	83	0.70	243
BR09	O	671	217	68	T	TSHE/BENE	29.6		91	0.77	294
BR17	W	853	225	73	T	TSHE/BENE	28.7	23	98	0.65	474
BR17	O	853	230	77	T	TSHE/BENE	28.7		72	0.85	66
DE06	W	640	143	30	M	TSHE/BENE	31.1	17	101	0.79	209
DE06	O	640	153	40	M	TSHE/BENE	31.1		80	0.87	172
LO01	W	731	128	62	T	TSHE/BENE–GASH	25.9	37	75	0.51	11
LO01	O	731	120	55	T	TSHE/BENE–GASH	25.9		74	0.80	184
LO05	W	701	102	28	T	TSHE/BENE	30.2	53	95	0.40	130
LO05	O	701	110	17	T	TSHE/BENE	30.2		103	0.78	201
LO06	W	701	288	50	T	TSHE/BENE	34.7	10	96	0.49	217
LO06	O	701	282	35	T	TSHE/BENE	34.7		102	0.74	493
LO07	W	518	155	10	T	TSHE/BENE–GASH	27.4	13	105	0.58	605
LO07	O	518	142	17	T	TSHE/BENE–GASH	27.4		102	0.90	617
LO08	W	792	124	60	T	TSHE/BENE–GASH	30.5	10	96	0.67	458
LO08	O	792	112	57	T	TSHE/BENE–GASH	30.5		98	0.89	231
MC03	W	701	220	32	T	TSHE/BENE–GASH	33.8	3	98	0.92	306
MC03	O	701	230	32	T	TSHE/BENE–GASH	33.8		84	0.68	162
MC04	W	671	180	33	M	TSHE/BENE–GASH	32.9	10	106	0.72	103
MC04	O	671	193	25	M	TSHE/BENE–GASH	32.9		110	1.02	255
SH09	W	671	181	53	T	TSHE/BENE	27.1	7	93	0.94	187
SH09	O	671	172	70	T	TSHE/BENE	27.1		95	1.05	97
SH13	W	640	194	30	M	TSHE/RHMA–BENE	27.1	13	93	0.63	238
SH13	O	640	199	30	M	TSHE/RHMA–BENE	27.1		94	0.75	203

<sup>a</sup>W, plot with residual trees; O, plot without residual trees.

<sup>b</sup>Topographic position: T, top one third of slope; M, middle one third of slope; B, bottom one third of slope.

<sup>c</sup>Plant associations are according to Hemstrom et al. (1987). TSHE, *Tsuga heterophylla*; BENE, *Berberis nervosa*; RHMA, *Rhododendron macrophyllum*

D. Don ex G. Don; GASH, *Gaultheria shallon*.

<sup>d</sup>King's (1966) 50-year Douglas-fir site index in metres; site trees drawn from both W and O plots at each location.

<sup>e</sup>Live residual trees per hectare.

<sup>f</sup>Age of oldest tree in younger cohort.

<sup>g</sup>Relative density of the younger cohort, computed as stand density index (SDI) divided by maximum possible SDI (SDI<sub>max</sub>) (Long 1985). SDI<sub>max</sub> was computed as the basal area weighted average of SDI<sub>max</sub> values for Douglas-fir and western hemlock listed by Long (1985).

<sup>h</sup>Volume of downed logs (m<sup>3</sup>/ha).

of residual trees at the time of initiation of the younger cohort. All cores were taken at breast height.

Dead trees that were measured included residuals of decay classes 1, 2, and 3 (Maser et al. 1988) and all dead younger cohort trees. For all these trees, we recorded decay class; total heights were recorded for all dead residual trees.

Segments of fallen trees (i.e., logs) within detection plots were measured. Species, decay class (following Maser et al. 1988), end diameters, and lengths were recorded. Logs smaller than 10 cm in diameter were excluded.

### Data summary

Volume of younger cohort trees was computed by different methods for conifers and hardwoods. Conifer volumes were computed following the tarif system, a method for selecting volume – basal area relationships for individual stands based on subsamples of trees with measured heights and diameters (Brackett 1973; Chambers and Foltz

1980; Avery and Burkhart 1994). For each unique combination of conifer species and plot, we selected all trees with measured heights. Volume was computed for these trees using the following equation:

$$[1] \quad \text{Volume} = b_0(\text{DBH}^{b_1})(\text{height}^{b_2})$$

Coefficient values were obtained from Browne (1962), with the exception of incense cedar (*Calocedrus decurrens* (Torr.) Florin.). For this species an equation was used that was developed from optical dendrometer measurements archived by the Andrews Experimental Forest Long-term Ecological Research program (Andrews LTER) (see Michener et al. 1990, data set “AND001”). From volume and DBH we computed the tarif number for each tree, then calculated the average tarif number for the species and plot. Average tarif numbers were computed separately for each plot (i.e., separately for each plot location and for plots with and without residual trees). For all live trees, species volume and plot volume was then computed from DBH and average tarif number (Brackett 1973).

Hardwood volumes were computed using equations from Pillsbury

**Table 2.** Attributes of the residual stand component.

Plot	Trees/ha		Basal area (m <sup>2</sup> /ha)		Live volume (m <sup>3</sup> /ha)	Crown area (m <sup>2</sup> /ha)	Percent with broken and (or) dead tops	Median DBH, cm (sample size)	
	Initial*	Live	Initial*	Live				Initial*	Live
BR07	56.7	33.3	32.3	24.9	376.3	1183	60	79.1 (9)	90.5 (10)
BR08	63.3	56.7	12.8	23.8	345.2	2305	12	35.1 (13)	65.0 (17)
BR09	26.7	13.3	29.1	15.7	269.5	1206	75	128.5 (3)	123.5 (4)
BR17	33.3	23.3	31.9	26.8	475.8	1054	57	116.1 (6)	124.0 (7)
DE06	16.7	16.7	7.8	13.9	213.8	1659	20	100.8 (2)	98.0 (5)
LO01	36.7	36.7	15.5	26	447.3	3078	36	60.2 (10)	83.0 (11)
LO05	53.3	53.3	20.9	48.3	836.7	5288	31	68.7 (9)	110.5 (16)
LO06	10	10	17.6	23	481.6	1595	0	158.4 (3)	180.0 (3)
LO07	16.7	13.3	22.3	19.5	329.6	1679	25	136.3 (3)	143.5 (4)
LO08	10	10	15.7	19	356.0	1298	100	143.1 (3)	160.0 (3)
MC03	3.3	3.3	3.5	4.6	101.7	287	0	115.6 (1)	132.0 (1)
MC04	10	10	7.1	9.6	168.6	1030	67	111.0 (3)	121.0 (3)
SH09	6.7	6.7	4.9	7	103.5	670	50	85.7 (1)	115.0 (2)
SH13	20	13.3	22.3	17.7	313.7	1125	100	112.2 (4)	122.5 (4)

\*Estimated live residual trees per hectare, basal area, and DBH at time of initiation of younger cohort.

and Kirkley (1984) and Snell and Little (1983). For individual hardwood trees lacking measured heights, height was estimated from diameter using the appropriate, species-specific equation from Garman et al. (1995).

Mean annual increment (MAI) of the younger cohort was computed as the total younger cohort volume on a plot divided by younger cohort age. Younger cohort age was defined as the age of the oldest tree in the younger cohort, converted from age at breast height to total age by adding King's (1966) correction factor for site class 3 (i.e., 8 years).

For the residual component of the stands, we estimated live trees per hectare and live basal area per hectare at the time of initiation of the younger cohort. Initial trees per hectare was defined as live residual trees, plus snags of decay classes 1, 2, and 3. Our observations of snags more advanced in decay suggested that they died during or prior to the event that initiated the younger cohort. Estimation of initial residual basal area was based on increment core data from residual trees. However, not all residual trees had cores of sufficient duration. For residual trees with complete increment data, DBH at younger cohort initiation was computed by subtracting twice the appropriate amount of radial increment (i.e., the amount of diameter corresponding to the age of the younger cohort) from current DBH. About 35% of the trees had partial ( $\leq 50$  year) or missing cores (the latter including snags of decay classes 1, 2, and 3). From the trees with complete cores, regression or ANOVA models were developed to estimate the missing data. For trees with partial increments, a linear regression model of first 50-year increment predicted by the most recent 50-year increment was acceptable ( $F = 188.5$ ,  $p < 0.0001$ ,  $R^2 = 0.75$ ,  $RMSE = 1.70$ ,  $n = 63$ ). The relationship between 100-year increment and DBH was weak. However, mean 100-year increment of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) was significantly different between plots (analysis of variance:  $F = 6.7$ ,  $p < 0.0001$ ,  $MSE = 19.6$ ,  $n = 41$ ). Thus, Douglas-firs without increment data were assigned the mean 100-year increment for the plot. The one residual western hemlock without a measured 100-year increment was assigned the mean increment of the other seven residual hemlocks.

To determine the amount of increment to subtract for snags, time since death was estimated as  $M_d$ , defined as the midpoint of range of age at which each decay class is reached for trees  $> 48$  cm (from Maser et al. 1988);  $M_d = 3$ , 13, and 35 years for decay classes 1, 2, and 3, respectively.

The many residual trees with broken tops complicated estimation of live volume of the residual component of the stands. Since the form

of the taper equation developed by Kozak (Avery and Burkhart 1994) provides for correction for volume in the missing top, it was used to estimate the volume of all live residual trees (i.e., intact as well as broken):

$$[2] \quad \text{Volume} = 7.85 \times 10^{-5} (\text{DBH})^2 (b_0 H_1) + (b_1/2) (H_1^2/H) + (b_2/3) (H_1^3/H^2)$$

where  $H$  is the intact height of tree, and  $H_1$  is the current height of tree. Species-specific coefficients developed from the Andrews LTER Experimental Forest dendrometer data (S. Garman, personal communication) were used (sample sizes from 53 to 420 trees per species;  $R^2$  values from 0.85 to 0.98). For trees with broken tops, it was necessary to estimate the intact height. These heights were estimated from DBH using the appropriate equations from Garman et al. (1995).

Crown area of residuals was computed from the average of the four crown radii, assuming circular crowns. Portions of crowns computed to fall outside of the residual plot boundary were excluded from the calculation.

#### Data analysis

To exploit the paired-plot design, response variables were computed as the percent difference of a given measurement in a plot with residual trees compared with the same measurement in the paired plot without residual trees. Using younger cohort volume as an example:

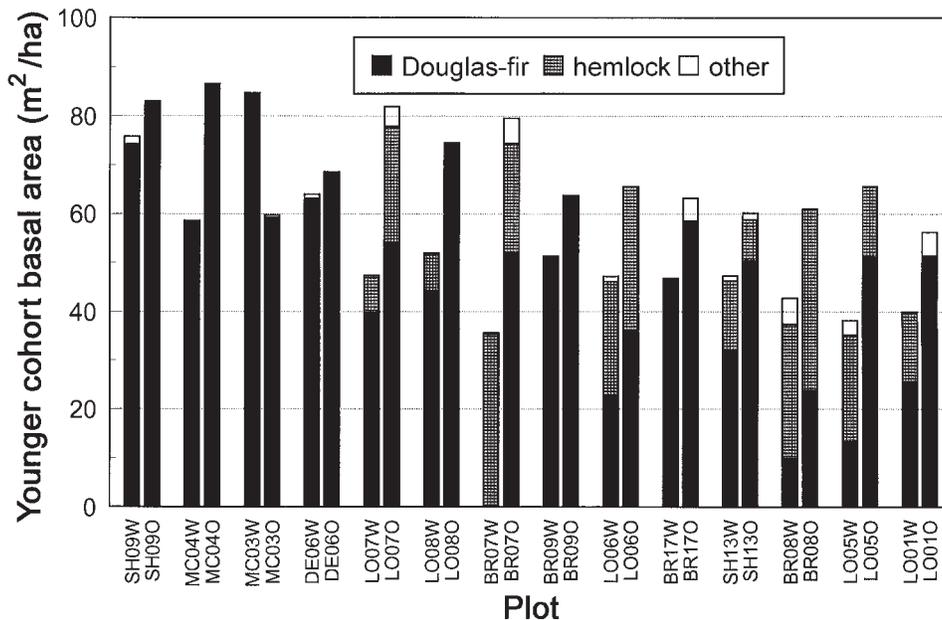
$$[3] \quad D\%_{\text{VOL}_i} = [(\text{VOL}_{\text{TW}} - \text{VOL}_{\text{IO}}) / \text{VOL}_{\text{IO}}] 100$$

where  $D\%_{\text{VOL}_i}$  is the the difference in percent of younger cohort volume for plot pair  $i$ ;  $\text{VOL}_{\text{TW}}$  is the younger cohort volume in plot  $i$  with residual trees; and  $\text{VOL}_{\text{IO}}$  is the younger cohort volume in plot  $i$  without residual trees.

One objective of the analysis was to identify the predictor variable that was most highly correlated with each response variable. Then each response-predictor pair was evaluated for the applicability of linear regression (SAS Institute Inc. 1989). For models that were statistically significant ( $p < 0.05$ ), the validity of the underlying assumptions of linearity, equal error variance, and normality of residuals was evaluated (SAS Institute Inc. 1989; Sabin and Stafford 1990).

Nonlinear responses of younger cohort basal area and volume to residual tree density have been suggested (e.g., Birch and Johnson 1992; Rose 1993). To test this possibility, various curvilinear models were evaluated in addition to the linear model. The curvilinear models fit to the data were

**Fig. 1.** Basal area of younger cohort trees per hectare in paired plots. Plot labels ending in W indicate plots with residual trees; plot labels ending in O indicate plots without residual trees.



$$[4] \quad Y = a + b(1/X)$$

$$[5] \quad 1/Y = a + bX$$

$$[6] \quad Y = a + bX + cX^2$$

These models allow for either a decreasing effect per residual tree (e.g., Birch and Johnson 1992) or an increasing effect per residual tree (e.g., Rose 1993), accommodate negative or zero response variable values, and accommodate the potential for values of zero for predictor variables. The linear model and the various curvilinear models were compared on the basis of  $R^2$  and the goodness-of-fit criteria.

In all cases, regressions were not forced to pass through the origin. Although it follows from our study design that there would be no difference between paired plots in the absence of residual trees, several problems would be caused by forcing regressions to pass through the origin. Regression through the origin alters the meaning of  $R^2$  and other diagnostic statistics (L. Ganio, personal communication). The confidence interval for predictions from regression through the origin is minimized at a value of zero for the predictor variable (Neter et al. 1983), unlike standard regression for which the confidence interval is minimized for the mean value of the predictor (Snedecor and Cochran 1980). Thus in this study, regression through the origin could decrease the precision of predictions in the range of residual tree values for which we had the most data. Declining to force regressions through the origin avoided these problems and kept the focus of the analysis on the range of residual tree values actually observed.

## Results

### Stand characteristics: residual component

The residual component of the stands ranged from a small fraction to more than half of values expected for mature or old-growth stands. Live basal area ranged from about 5 to 48 m<sup>2</sup>/ha, compared with average values for mature and old-growth Douglas-fir forest in western Oregon and Washington of 59 and 69 m<sup>2</sup>/ha, respectively (Spies and Franklin 1991) (Table 2). Live volume of sampled stands, including top and stump, ranged from about 100 to almost 840 m<sup>3</sup>/ha. For the

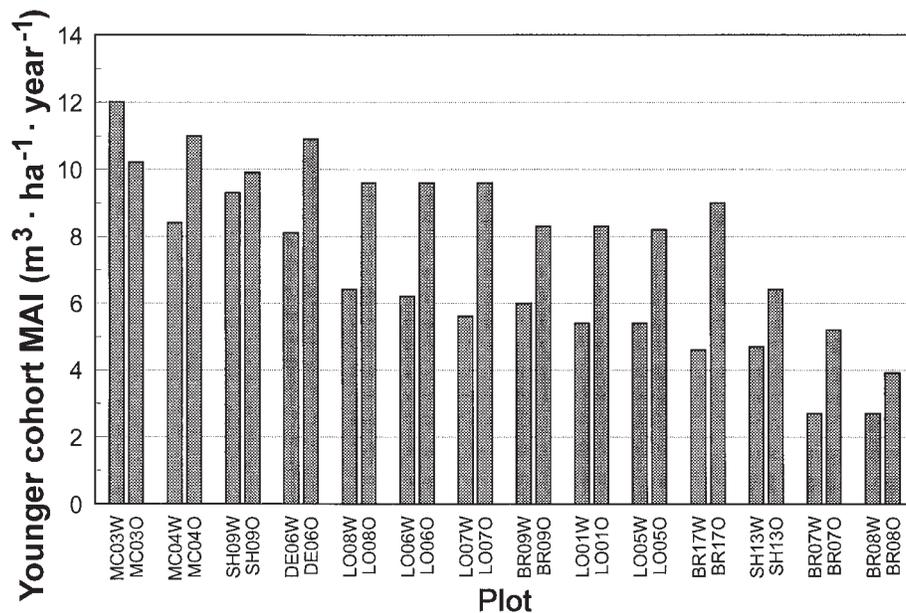
predominantly mature and old-growth plots that Hemstrom et al. (1987) used to characterize the plant associations of our plots, average volume ranged from about 950 to 1150 m<sup>3</sup>/ha. Live crown area ranged from about 300 to 5300 m<sup>2</sup>/ha. Damaged tops were common among the residual trees; over all the plots 39% of residual trees had broken and (or) dead tops. Median DBH of live residual trees was 103 cm. For the live residual trees with usable increment cores (78% of the total), the median estimated DBH at the time of initiation of the younger cohort was 87.6 cm. Nearly all the residual trees were Douglas-fir (90% of stems and 97% of basal area); the remainder were western hemlock (9% of stems and 2.6% of basal area) and western red cedar (*Thuja plicata* Donn.).

### Stand characteristics: younger cohort

Most younger cohort trees were Douglas-fir or western hemlock. Over all the plots, Douglas-fir accounted for 59% of stems and 81% of basal area; western hemlock accounted for 37% of stems and 17% of basal area. Douglas-fir was more ubiquitous than western hemlock; Douglas-fir was in all but one plot, whereas hemlock was absent from 7 of the 28 plots. Other species present included three conifers (western red cedar, incense cedar, and western yew (*Taxus brevifolia* Nutt.)), and four hardwood species (golden chinkapin (*Castanopsis chrysophylla* (Dougl.) DC), bigleaf maple (*Acer macrophyllum* Pursh), Pacific dogwood (*Cornus nuttalli* Aud.), and Pacific madrone (*Arbutus menziesii* Pursh)).

For the most part, plots without residual trees had more younger cohort trees, basal area, and volume than the paired plots with residual trees. In 10 of the 14 pairs, there were more younger cohort trees in plots without residual trees. In 13 of 14 plots, younger cohort basal area (Fig. 1) and volume were greater in the plot without residuals. Total live volume of both residual and younger cohort trees on plots with residual trees ranged from about 700 to about 1350 m<sup>3</sup>/ha. Total live

**Fig. 2.** Mean annual increment of younger cohort trees in paired plots. Plot labels ending in W indicate plots with residual trees; plot labels ending in O indicate plots without residual trees.



**Table 3.** Pairwise Pearson correlation coefficients between predictor and response variables.

	RBA <sub>I</sub>	RBA <sub>L</sub>	RCRAR	RTPHI	RTPHL	RVOLL
D% <sub>BA</sub>	-0.61 (0.02)	-0.62 (0.02)	-0.44 (0.12)	-0.55 (0.04)	-0.48 (0.08)	-0.57 (0.03)
D% <sub>HT</sub>	-0.53 (0.05)	-0.17 (0.56)	-0.05 (0.87)	0.00 (0.99)	0.15 (0.60)	-0.23 (0.43)
D% <sub>MAI</sub>	-0.76 (0.002)	-0.63 (0.02)	-0.34 (0.23)	-0.53 (0.05)	-0.42 (0.13)	-0.60 (0.02)
D% <sub>TPH</sub>	0.08 (0.79)	-0.46 (0.10)	-0.46 (0.10)	-0.36 (0.21)	-0.47 (0.09)	-0.44 (0.12)
D% <sub>VOL</sub>	-0.71 (0.005)	-0.62 (0.02)	-0.40 (0.16)	-0.52 (0.06)	-0.42 (0.13)	-0.59 (0.03)
D% <sub>VOLT</sub>	-0.04 (0.90)	0.48 (0.08)	0.51 (0.06)	0.38 (0.18)	0.53 (0.05)	0.48 (0.08)

**Note:** Probabilities are given in parentheses. Sample size is 14 in all cases. D%<sub>BA</sub>, difference in percent of younger cohort basal area between paired plots; D%<sub>HT</sub>, difference in percent of average height of younger cohort dominant and codominant trees; D%<sub>MAI</sub>, difference in percent of younger cohort MAI; D%<sub>TPH</sub>, difference in percent of younger cohort trees per hectare; D%<sub>VOL</sub>, difference in percent of younger cohort volume; D%<sub>VOLT</sub>, difference in percent of younger cohort plus residual live volume. RBA<sub>I</sub>, estimated basal area of live residual trees at initiation of younger cohort; RBA<sub>L</sub>, basal area of live residual trees; RCRAR, crown area of residual trees; RTPHI, estimated live residual trees per hectare at initiation of younger cohort; RTPHL, live residual trees per hectare; RVOLL, volume of live residual trees.

volume was higher on plots with residual trees for 11 of the 14 pairs of plots. Younger cohort MAI ranged from 2.7 to 12.0 m<sup>3</sup>·ha<sup>-1</sup>·year<sup>-1</sup> and was greater without residual trees for 13 of the 14 pairs of plots (Fig. 2). Although mean height of dominant and codominant conifers varied substantially between plots, there was not a consistent difference between paired plots. Means varied from about 24 to 43 m.

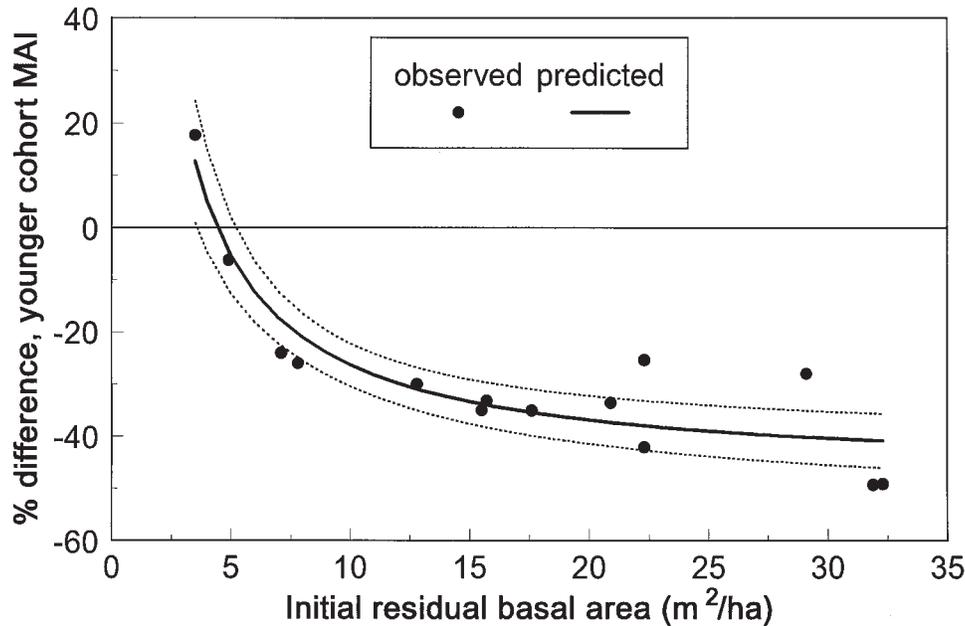
From the sample of younger cohort trees that were cored, the period of recruitment varied markedly between plots. The range of ages of younger cohort trees varied from 9 to 67 years (samples sizes from 6 to 22). For the most part, the age range was similar within pairs of plots.

### Tests of effects of residual trees

The pairs of responses and predictors selected for regression analysis (Table 3) were

- (1) percent difference between paired plots in younger cohort basal area and basal area of live residual trees;
- (2) percent difference in average height of younger cohort dominant and codominant trees and estimated live basal area of residual trees at time of initiation of the younger cohort;
- (3) percent difference in younger cohort MAI, and estimated live basal area of residual trees at time of initiation of the younger cohort;

**Fig. 3.** Percent difference in the younger cohort mean annual increment versus estimated initial residual basal area, with regression line ( $Y = -47.4 + 210.2(1/X)$ ,  $R^2 = 0.85$ ) and 95% CI for predicted mean response.



**Table 4.** Results of tests of effects of residual trees using linear regression.

Response <sup>a</sup>	Predictor <sup>a</sup>	Result	Model	R <sup>2</sup>	RMSE
D% <sub>BA</sub>	RBA <sub>L</sub>	$p = 0.0001$	$D\%_{BA} = -49.5 + 369.4(1/RBA_L)$	0.73	12.3
D% <sub>HT</sub>	RBA <sub>I</sub>	ns <sup>b</sup>			
D% <sub>MAI</sub>	RBA <sub>I</sub>	$p = 0.0001$	$D\%_{MAI} = -47.4 + 210.2(1/RBA_I)$	0.85	6.9
D% <sub>TPH</sub>	RTPHL	ns			
D% <sub>VOL</sub>	RBA <sub>I</sub>	$p = 0.0001$	$D\%_{VOL} = -50.0 + 265.7(1/RBA_I)$	0.83	9.3
D% <sub>VOLT</sub>	RTPHL	ns			

**Note:** Sample size was 14 for all significant models.

<sup>a</sup>Variable names are as in Table 3.

<sup>b</sup>Linear regressions with untransformed or transformed data not significant (i.e.,  $p > 0.05$ ).

- (4) percent difference in younger cohort trees per hectare and live residual trees per hectare;
- (5) percent difference in younger cohort volume and estimated live basal area of residual trees at time of initiation of the younger cohort; and
- (6) percent difference in younger cohort plus residual live volume and live residual trees pre hectare (see Table 3 for abbreviations).

For most of the response variables, several predictors had nearly the same degree of correlation. Thus, for many of the response variables, alternative predictors may have been nearly as powerful.

Regressions of all the response variables pertaining to younger cohort basal area and volume (i.e., D%<sub>BA</sub>, D%<sub>MAI</sub>, D%<sub>VOL</sub>) on residual basal area were significant (Table 4). All of these response variables declined with increasing residual basal area. In all cases, the most appropriate model was curvilinear (i.e., 1/X; Table 4); for all these models the decline in the younger cohort response was most pronounced at low levels of residual basal area (e.g., Fig. 3). As judged by R<sup>2</sup> values, these models fit the data well (R<sup>2</sup> ranged from 0.73 to 0.85). For the remaining response variables, regression with linear or curvilinear models was not significant.

To summarize the results of the regression analysis, predicted mean values and corresponding 95% confidence intervals (95% CI) were computed for all significant models, assuming initial residual basal area of 10 m<sup>2</sup>/ha (approximately the current green tree retention levels on matrix land for federal forests in the range of the northern spotted owl (USDA Forest Service and USDI Bureau of Land Management 1994)). The mean percent difference in younger cohort basal area was -26% (95% CI: -33% to -19%), the mean percent difference in younger cohort MAI was -26% (95% CI: -30% to -22%), and the mean percent difference in younger cohort volume was -23% (95% CI: -29% to -18%).

## Discussion

### Effects of residual trees

We used stands within the western hemlock zone in which the quantity of residual trees was similar to recent prescriptions for green tree retention on federal forests in western Oregon (i.e., 5–25 trees/ha; Birch and Johnson 1992). Residual trees had a negative effect on MAI, standing volume, and basal area of the younger cohort. Basal area of residual trees was more

strongly correlated with these younger cohort variables than were other measures of the residual component of the stands (e.g., trees per hectare, volume). For these stands, the effect of residuals on younger cohort MAI, basal area, and volume can be represented by curvilinear relationships, with the effect per unit residual basal area decreasing as residual basal area increases. As an example, at 10 m<sup>2</sup>/ha residual basal area, we predicted a decrease of 26% in younger cohort MAI. This amount of residual basal area could be made up of 13 residual trees of 100 cm DBH, 23 residual trees of 75 cm DBH, or 51 residual trees of 50 cm DBH per hectare. Residual trees did not show an effect on volume of the total stand, younger cohort trees per hectare, or average height of younger cohort dominant and codominant trees.

Simulation modeling studies provide perhaps the only results for comparison with this study. Simulation experiments have been conducted to assess implications of green tree retention on sites in the western hemlock zone similar to ours but with planting and thinning of trees similar to current management of federal forests in western Oregon (Birch and Johnson 1992, Hansen et al. 1995). Comparison between the simulation studies and our results are complicated by differences in predictor and response variables and, for one study, retention of trees smaller (quadratic mean diameter of 55 cm) and presumably younger than the residual trees in this study (Birch and Johnson 1992). However, general trends in the simulation studies were similar to our results. Hansen et al. (1995) reported cumulative basal area (basal area of harvested trees plus trees standing at the end of the 240-year simulation period), with retention of 0–150 trees/ha from an old-growth stand and rotation lengths of 40, 80, 120, or 240 years. For all rotation lengths, they found a substantial drop in cumulative basal area between 0 and 5 retained trees/ha (e.g., 25% drop for 80-year rotation) and a more gradual decline for additional increases in retention (e.g., 10% drop between 5 and 20 retained trees/ha, 80-year rotation). Birch and Johnson (1992) reported net volume growth of both retained trees and the younger cohort, after one or two rotations with retention of 0–49 trees/ha from a mature stand, using rotation lengths of 60 or 90 years. They also found the greatest decline per retained tree occurred at low retention densities. With a 60-year rotation, net volume growth dropped 18% with 11 trees/ha retained and 25.5% for 49 trees/ha retained. Results were similar for a 90-year rotation.

The significant regression models predicted a positive effect of very low amounts of residual trees, resulting from the single pair of plots that showed such an effect. Until more information is accumulated to confirm such effects, we are doubtful that there is an important positive effect on younger cohorts of low levels of residual trees. Because of the use of curvilinear models and the good fit of these models to the data (e.g., Fig. 3), inclusion of this observation had little effect on predictions for higher amounts of residual trees.

### Comparison to managed stands

The stands in this study differed from current managed stands with respect to the density of the younger cohort, the interval of tree re-establishment, and downed coarse woody debris. In addition to potential biological differences between natural and managed stands, our small plot size relative to the spatial scale of management activities may also be significant.

Compared with a “typical” managed Douglas-fir plantation under the Willamette National Forest Plan (USDA Forest Service 1990; J. Mayo, personal communication), the stands in our study had quite high tree densities. Standard management includes planting trees at about 1400/ha, thinnings at ages 12, 47, and 67, and assumes natural mortality of about 6 trees·ha<sup>-1</sup>·decade<sup>-1</sup> thereafter. In comparison with this scenario, stands of similar age in our study had densities between two and seven times as high as managed stands. Several of the plots had densities approaching or exceeding the planting density for managed stands, at ages up to 140 years. The high density of our stands is also reflected in the relative density values for the plots without residual trees (Table 1). These plots had relative densities between 0.68 and 1.05, much higher than target densities of 0.3–0.6 for managed Douglas-fir stands in the Pacific Northwest (Tesch 1995).

That the stands in this study were quite dense compared with managed stands suggests that, with density control, it may be possible to limit the decrease in volume growth under green tree retention. Also, at lower densities seedlings need not be planted close to retained green trees. Another factor relating to density control in managed stands is that the ratio of lumber volume to total volume tends to increase with increasing tree size (Avery and Burkhart 1994). Thus, if a measure of volume is used that pertains more directly to lumber yield, management to limit density of tree regeneration in areas subject to green tree retention may mitigate volume losses even more than would be apparent from comparison of total volumes.

In contrast to contemporary legal and policy requirements for speedy regeneration of trees after timber harvest, the stands in this study exhibited a long period of tree recruitment. The Willamette National Forest Plan (USDA Forest Service 1990) requires establishment of a new tree stand within 5 years after harvest in most cases. In our stands, ages of cored individuals varied as much as 67 years within a plot (minimum 9 years, average 28 years). It is likely that this extended period of recruitment will decrease the stand-level rate of wood volume accumulation because of the extended period before full site occupancy. On the other hand, recruitment of trees over several decades may increase the diversity of tree sizes and species.

Whereas intensive management for wood production has until recently included removal of dead wood (Franklin et al. 1986), natural stands of all successional stages generally contain considerable amounts of downed logs and snags (Spies et al. 1988; Hansen et al. 1991). With respect to log volume, stands in this study were similar to reported values for natural young stands (average volume 242 m<sup>3</sup>/ha, range 11 to 617, see Table 1; average for natural young stands was 248 m<sup>3</sup>/ha in Spies et al. 1988). Though future management of federal forests will include leaving more dead wood (USDA Forest Service 1990; USDA Forest Service and USDI Bureau of Land Management 1994), amounts may continue to be lower than averages for natural stands (e.g., minimum log volume prescribed in USDA Forest Service and USDI Bureau of Land Management (1994) for Willamette and nearby National Forests is about 40 m<sup>3</sup>/ha). Decaying wood can serve as a source of organic matter and nutrients for forest soils (Harmon et al. 1986, Maser et al. 1988). However, long-term analyses to confirm or refute an effect of timber harvest and removal of coarse

woody debris on forest productivity in the western hemlock zone are apparently lacking.

This study quantified the younger cohort of two-aged stands within 500 m<sup>2</sup> plots, but management activities such as green tree retention are implemented over much larger areas. Since our data are best fit by nonlinear models (i.e., 1/X), the predicted effect on younger cohort basal area or volume over a larger area cannot necessarily be estimated from the mean residual basal area for an entire area because of the potential for uneven distribution of retained trees. In particular, integrating predictions for the younger cohort over a larger area would probably require predictions for localized areas without residual trees, or with low values of residual basal area (e.g., ≤5 m<sup>2</sup>/ha). However, low values of residual basal area were not well represented in our data set. Thus, dynamics of two-aged stands with relatively few residual trees remains an important area for investigation. In addition, large management areas may encompass more variability in site conditions than the relatively narrow range of plant associations that we examined.

### The retrospective approach

Retrospective studies of forest dynamics require assumptions about stand history. Those assumptions should be borne in mind in interpreting our results and also may suggest worthwhile research. Especially important are assumptions concerning the disturbances that initiated the younger cohorts, and the dynamics of residual trees.

To infer that the differences between plots with and without residual trees were due to the residual trees we had to assume that other effects of the disturbance that initiated the younger cohorts were not correlated with survival of residual trees. However, fire can affect all aspects of forest ecosystems, not just dominant trees (Agee 1993). For example, the local variations in tree mortality that left the scattered residual trees in our stands may have corresponded to patterns in intensity of burning of the forest floor. Since the degree of soil disturbance can influence succession (Dyrness 1973; Halpern 1988), some of the observed patterns could be due to fire effects other than tree mortality. Contemporary wildfires in the western hemlock zone present opportunities to test directly the degree to which canopy tree mortality is correlated with soil disturbance, and the effects of both on forest growth and succession. In addition, these questions should be addressed in studies of actual green tree retention harvest units.

Some important aspects of the dynamics of the residual trees were either assumed or not addressed. We assumed that the time of death of snags could be inferred from the estimated time it takes trees of different sizes to decay to different extents (i.e., decay classes; Maser et al. 1988). This is an oversimplification of the dynamics of decay of individual snags. We observed that many of the residual trees had broken and (or) damaged tops and so could not estimate past height or volume growth of residual trees. Better understanding of timing of top damage or dieback, timing of tree death, and patterns of height growth of residual trees following disturbance is critical for quantifying both the effects of residual trees on younger cohort growth and other ecosystem processes, and the contribution of residual trees to total stand volume growth. Long-term studies of residual trees are needed, both on recent wildfires and on operational green tree retention units.

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