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# Dynamics of hardwood patches in a conifer matrix: 54 years of change in a forested landscape in Coastal Oregon, USA

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## Abstract

Changes to minor patch types in forested landscapes may have large consequences for forest biodiversity. The effects of forest management and environment on these secondary patch types are often poorly understood. For example, do early-to-mid successional minor patch types become more expansive as late successional forest types are fragmented or do they also become more fragmented in managed landscapes? We evaluated the dynamics of early-to-mid successional hardwood patches in a conifer-dominated landscape in relation to environment and land ownership in the central Coast Range of Oregon, USA, from the time of early logging to the present-day using scanned and georeferenced aerial photographs and a GIS. Hardwood patches declined in size, number, total area, and within-patch cover-type heterogeneity, and became more irregular in shape. Patch turnover and fragmentation was high, with most patches present at the historical date disappearing by the present-day. Land ownership was important to hardwood patch dynamics: hardwoods declined on lands owned by the USDA Forest Service, increased on non-industrial private lands, and were at similar levels at both dates on private forest industry lands. Patch locations became more restricted to near-stream, lower elevation areas where hardwoods are competitive. The relatively extensive distribution of hardwood patches at the historical date probably resulted from earlier fire, selective logging, and grazing. In recent decades, forest management that includes fire suppression and intensive management, and ecological constraints have resulted in a landscape in which early-to-mid successional hardwood patches have been reduced in size, fragmented, and restricted to particular locales.
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*Keywords:* Patch population dynamics; Red alder (*Alnus rubra*); Historical landscapes; Oregon coast range; Forest management

## 1. Introduction

Hardwood trees are important sources of biological diversity in many of the world's conifer-dominated forests (Essen et al., 1997; Mladenoff and Pastor, 1993; Van Cleve, 1986). Within these forests, patches of hardwoods are important sources of species, nutrients, and microclimatic diversity (Ellenberg, 1988; Fried et al., 1990; Johnson and Freedman, 2002; Perry et al., 1987; Romanya and Vallejo, 1996). The relationships of the spatial pattern and dynamics of hardwood patches to environmental and human influences, such as forest management, are not well understood. Forest management is typically associated with fragmentation of large late-successional patches into smaller more isolated fragments (Mladenoff et al., 1993; Spies et al., 1994) but we lack empirical information on what happens to early-to-mid successional patches of hardwoods in managed landscapes. Do patches of hardwoods expand or become more fragmented under forest management and how does management intensity affect the trend?

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Forman (1995) states that human disturbances can either increase or decrease patch shape complexity depending on management intensity, and that topography will also constrain patch shape. Shifts in the amount, size, shape, or location of minor patch types such as hardwoods in a conifer matrix may be associated with changes to a landscape's resistance or resilience to disturbance and declines in biodiversity. Studies of these changes in pattern and related processes are needed if we are to make better predictions of how forest mosaics and biological diversity will change in different management and environmental settings.

In Pacific Northwestern North America, hardwood tree patches, typically dominated by red alder (*Alnus rubra*) or bigleaf maple (*Acer macrophyllum*), are important to the rate and direction of forest succession and regeneration following disturbance (Hemstrom and Logan, 1986). They are hotspots for native biodiversity (Neitlich and McCune, 1997), providing food and cover for a variety of mammals, amphibians, invertebrates, and birds (McComb, 1994), and are an important substrate for arboreal lichens and mosses (Peterson and McCune, 2001). Hardwood patches are an important site of canopy epiphytes and invertebrates within the conifer matrix and can serve as an important source of available nitrogen in the nitrogen-limited forest ecosystems of the region. Hardwoods are associated with greater macroinvertebrate biomass in streams (Piccolo and Wipfli, 2002). Hardwoods enhance soil stability and soil and stream fertility in Pacific Northwest forests through N-fixation, production of high N-content litter, rapid regeneration after disturbance, and persistence in flood-prone and disturbed areas (Berg and Doerksen, 1975; DeBell, 1990; Fried et al., 1990; Haeussler et al., 1995; McDonald and Tappeiner II, 1990; Owston, 1990; Tarrant and Trappe, 1971). Consequently, obtaining a better understanding of hardwood patch dynamics, including changes in the size distribution of hardwood patches, patch shapes, within-patch cover type composition, and the spatial distribution of hardwood patches in relation to topography and ownership, can provide insights into ecosystem function and the potential habitat for many important components of biodiversity.

In the Oregon Coast Range landscape, timber harvest over the past fifty years has had a major impact on forest composition and configuration, with a reduction in older forest and old-forest patch size and an increase in old-forest patch fragmentation (Cohen et al., 2002; Ripple et al., 2000; Spies et al., 1994). It remains unknown how logging has affected hardwood tree patch patterns and dynamics. One opinion is that hardwood patch number and extent have increased from European settlement (mid-1800s) to present in response to catastrophic fire and logging (Carlton, 1988; Hibbs and Giordano, 1996; Spies et al., 2003). The extensive disturbance history of the Oregon Coast Range might be expected to lead to an increase in early-to-mid successional hardwoods. On the other hand, high intensity forest management in recent decades including broadleaf herbicide application and legally-required conifer regeneration might be expected to reduce hardwoods (Alig et al., 2000; Oregon Department of Forestry, 2001).

The purpose of this study was to investigate how hardwood patch conditions in present-day managed forests compare with those of post-settlement historical forests, and to discuss the implications of our findings with respect to ownership and environmental gradients. Our underlying assumption is that ecological processes and biodiversity patterns related to hardwoods would undergo modifications with shifts in hardwood patch configuration and amount. Our specific study objectives were to: (1) describe the change in hardwood patch number, size, shape, and internal cover-type composition in the central Coast Range landscape (total land area = ~280, 798 ha) from the late 1930s to the present; (2) compare current and early logging (1939) numbers and sizes of hardwood patches with regard to land ownership patterns and environmental gradients; and (3) track a population of hardwood patches over time to determine the dynamism of a single population by evaluating the persistence of the patches in the population and changes in their size, shape, and location. Such information would provide a valuable perspective on forest dynamics in western Oregon and provide reference information for evaluating hardwood patch dynamics in other regions.

## 2. Study area

The study was carried out in the central part of the Coast Range mountains of Oregon (from about 44.2–45.0°N and 123.7–124.1°W). The climate is maritime with cool, dry summers and warm, wet winters. Precipitation is high (2500–3000 mm/year) and occurs primarily as winter rains. Soils are marine sedimentary Tyee sandstones with sparse volcanic basaltic rock outcrops (Franklin and Dyrness, 1988). The terrain is steep and rugged, with slope gradients between 0° and 69°, elevations from sea level to 1102 m, and a dense network of intermittent and perennial streams. Smaller, lower-order streams tend to be located at higher elevations and have a higher incidence of constrained reaches than the larger streams.

The study area is in the western hemlock (*Tsuga heterophylla*) and Sitka spruce (*Picea sitchensis*) vegetation zones (Franklin and Dyrness, 1988). Dominant tree species include Douglas-fir (*Pseudotsuga menziesii*), western hemlock, western redcedar (*Thuja plicata*), and Sitka spruce, the latter restricted to the coastal strip. Hardwood trees (primarily red alder and secondarily bigleaf maple) are less common but occur in patches throughout the coniferous matrix, especially in recently disturbed and riparian areas (Franklin and Dyrness, 1988); in terms of longevity, red alder live 80–120 years and bigleaf maple 250–300 years. Hardwood trees have been present in the Coast Range at moderate levels since early in the Holocene, with a peak in abundance of red alder at 3500 years before present that coincided with a period of increased fire frequency, and a decrease at around 1950 years before present that coincided with increases in Douglas-fir and true fir (*Abies* spp.) and a decrease in fire frequency (Long et al., 1998). At present-day, vegetation occurs primarily in managed forest blocks of maximum age 60 years on private lands and 150 years on public lands. Land ownership is industrial private (44%), USDA Forest Service (31%), and non-industrial private (17%); the remaining 8% is under USDI Bureau of Land Management, state, and miscellaneous ownership. Land ownership by private industry and the Forest Service is not distinguished strongly by differences in topography, unlike other areas in the Pacific Northwest and other regions. Private non-industrial lands tend to be located along larger valley streams. Changes in land ownership in the past several decades have occurred primarily within and not between the three major ownership classes (Azuma et al., 2002).

Historically disturbance was dominated by high intensity stand-replacing fires with a return interval of 200–300 years (Agee, 1993). The most recent large fires in the study area occurred in 1850–1880, and fire suppression followed by the mid-1900s; post-fire, unmanaged forests are 130–150 years old and are in the mature or understory reinitiation stage of development (Impara, 1997; Oliver and Larson, 1990; Poage, 1994; Spies, 1997). Finer scale (<1 ha) natural disturbance has included windthrow events in the 1950s and 1960s, large floods in the 1960s, and chronic mortality of Douglas-fir and western redcedar by laminated root rot fungus patches (Orr, 1963; Ruth and Yoder, 1953; Spies and Cline, 1988; Strome, 1986). Historically, Native Americans frequently set low intensity fires on the coastal fringe and along the major river valleys (Hays, 1976). European settlement in the mid- to late-1800s brought land clearing, selective logging and sheep grazing. In the early part of the 1900s logging railroad lines and the selective high grade logging that accompanied them mainly followed the large river corridors. By the 1950s, road construction and clearcutting had become the most common disturbance types, and because of the development of the road network, clearcutting could occur throughout the landscape. On private lands, the allowed size of harvest units had no limit until the early 1990s, and large amounts of forest were cut on industrial private lands from 1930s to 1990s (Garman et al., 1999). On federal lands, where cutting began later, the staggered setting harvest method has scattered 10–20 ha clearcuts across areas of older forest since the late 1940s.

## 3. Methods

We examined differences between two randomly selected populations of hardwood patches separated by 54 years and tracked the historical, early-logging patch population to determine how modifications to individual patches occurred over this time interval. We located hardwood patches on 74 non-adjacent historical (1939; 1:27,000) panchromatic and the corresponding 160 current (1993; 1:12,000) color aerial photos. We selected photos from these dates because the 1939 set had the combined broadest extent and earliest date of coverage for the central Coast Range area, and the 1993 photos was taken just prior to a large flood event (in 1996) which is likely to have modified hardwood patch configurations and extents, especially in riparian areas, and our goal was to sample patches representative of the 50 + year time period as a whole. We scanned the photos so that they had similar grain sizes of 1.7 m (historical) and 1.5 m (current), both well below the average crown size of red alder (4.8–7.9 m) (Smith, 1968), the species which typically has the smallest crown size of the hardwood tree types. We georeferenced the digital photo images in ARC using a 1:100,000-scale streams coverage from the Environmental Protection Agency's River Reach Database and two roads coverages: the 1:100,000-scale US Census TIGER/Line roads file for 1995, and a Siuslaw National Forest roads coverage derived and updated in 1991 from Cartographic Features Files digitized from quadrangle maps and supplied by the Geometronics Service Center. These three coverages are considered to be of good to high accuracy and have been used in other Forest Service research in the Coast Range. We used the roads coverages in preference to the streams coverage whenever applicable because they are considered to be of slightly higher accuracy.

To obtain the locations of hardwood patches, we first determined the area shared by the two sets of digital photo images, and randomly sampled 1600, 20 m pixels on the photos to serve as potential starting points for locating patches. These were outlined by a 20 m transparent polygon grid superimposed on the entire photo coverage area in a geographic information system (GIS). The same 1600 pixels were sampled for both dates. For each photo date, we manually examined the finer-scale area of each digital photo within and around each of these 20 m pixels. We identified the cover type within each of 1508 pixels (those of the 1600 of adequate photo quality) as one of the following 14 types: bare soil, agriculture/pasture, shrub, hardwood tree, small mixed conifer, medium mixed conifer, large mixed conifer, very large mixed conifer, small conifer, medium conifer, large conifer, very large conifer, water and road. Those 20 m polygons that we classified as hardwoods were examined again on the photos in the next steps. Mixed and pure conifer size classes corresponded to dominant conifer crown diameters as follows: small (0.1–2.5 m), medium (2.6–5 m), large (5.1–10 m), and very large (>10 m). Mixed types had less than 50% coniferous cover. The 20 m pixel size outlining the sample unit corresponded to the scale of the vegetation; 20 m is the approximate diameter of 1 very large Douglas-fir tree crown and also the diameter of the crowns of a group of 2–3 red alder trees (a very small hardwood patch). Thus we evaluated each photo at its 1.5–1.7 m resolution within each 20 m pixel. We used a single photo interpreter (Kennedy) and ground-truthed our interpretation of a subset of photos to limit photo interpretation error (Haack, 1962).

We followed two approaches to delineate the two populations of hardwood patches. First, we started from each 20 m pixel identified as hardwood at each photo date. Working outward from each of these pixels across the photo in all directions including diagonals, we labeled the adjacent hardwood 20 m pixels in the polygon grid as members of the contiguous hardwood patch (Fig. 1). We also labeled as within the patch any non-hardwood inholdings such as shrubs or conifers that were completely surrounded by the contiguous hardwood patch, and assigned the appropriate cover type to those non-hardwood pixels. We considered diagonals as contiguous in patch definition to account for narrow or filamentous hardwood patches that commonly occur in riparian areas and roads. Our second approach to delineating hardwood patches was designed to include patches in the present-day population which consisted of any fragments of historical patches, and of any patches present in the historical population which may have shifted overall location through individual tree mortality or establishment which would not otherwise be accounted for in the current population using the first approach. Therefore, after we had delineated hardwood patches using the first method, we reexamined all the pixels comprising all the historical hardwood patches on current imagery, and for those pixels that we determined were still hardwood in 1993, we also delineated present-day patches outward from those pixels in the same manner as the first approach, on the current imagery. Thus we obtained two populations of patches separated by time and developed the information to track a single population over time.

We compared the populations of hardwood patches in historical and present-day forest landscapes for the following hardwood patch attributes: size (ha), shape, core to edge area ratio, and within-patch heterogeneity. We also evaluated historical patches for their degree of fragmentation (i.e., fragmented/coalesced/no change/disappeared) and their type of area change from 1939 to 1993 (i.e., increased/decreased/no change/disappeared). Weighted mean patch size was calculated as the mean area of all patches weighted by the sum of the inverse of individual patch areas. We obtained or derived shape, core to edge area ratios, and fragmentation classes from FRAGSTATS (McGarigal and Marks, 1995). Shape, an index used to quantify shape complexity, was equal to 0.25 times the length of the patch perimeter divided by the square root of the patch area. A shape value of

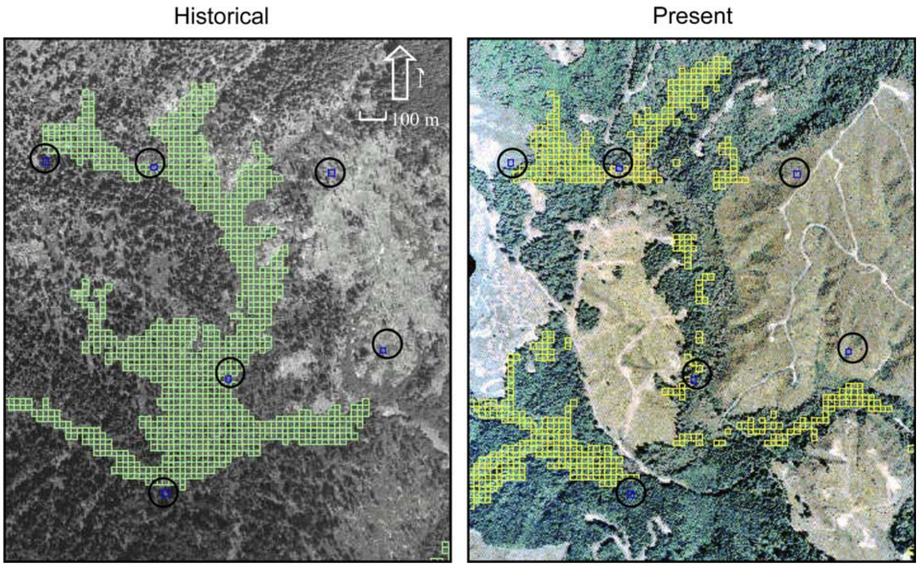

Fig. 1. Time series of change in a single historical (1939) hardwood patch that fragmented into multiple, smaller sub-patches by present-day (1993). Each square shown measures 20 m on a side. Those squares encircled are some of the 1600 squares that were cover-typed; the subset with adjoining squares were cover-typed as hardwood. A stream runs north-south through the center of both photos.

1 was square, and increasing values were more irregular in shape. We defined the edge as the outer 20 m band within each patch, and core area as any patch area that remained after the band area was subtracted. For each patch, we calculated within-patch heterogeneity for each non-hardwood cover type occurring within the patch by dividing the area of the non-hardwood cover type by the total patch area, and total within-patch heterogeneity by dividing the total area of non-hardwood cover types within each patch by the total patch area. For categorical fragmentation classes we evaluated the change in the number of sub-patches from tracked historical patches.

To obtain environmental and ownership attributes for each patch we overlaid each patch on ARC coverages and grids we acquired or created and obtained the mean value for the patch for each attribute. These attributes included: elevation (m), stream distance (m), distance to the ocean (km), slope (%), slope position (% of slope), topographic curvature, annual solar radiation (cal/(cm$^2$ day)), ownership (industrial private/Forest Service/non-industrial private/other), ecoregion (coastal lowlands/coastal uplands/coastal volcanics/volcanics/mid-coast sedimentary), stream order (first and second order/third and higher order), riparian class (riparian/upland), and valley class (flat/sloping). We derived slope position, which ranged from 1 (lower slope) to 100 (upper slope), from the 10 m resolution digital elevation model we used for elevation, using an ARC Macro Language and setting sink and peak fill limits to 90 cells and valley and ridge accumulation minimums to 1350 cells. We derived topographic curvature from the 10 m digital elevation model using ARC GRID commands. Annual solar radiation was calculated using monthly climate data from a network of weather stations in the study area and the program SolarImg (Harmon and Marks, 1995). We obtained both historical and current hardwood patch ownership attributes from a 1993-dated ownership coverage because most changes of ownership during the elapsed 54 years occurred within, and not between, the broad categories of ownership used in this study (Azuma et al., 2002). We designated riparian areas as areas within 60 m of streams, in accord with the riparian buffer width used by the Forest Service within the study area. We designated valley bottoms as within-riparian-buffer areas of little slope change, having slope change values of 1–10 in ARC CURVATURE.

We conducted statistical analysis in SAS. Student's *t*-tests were used to test for significant differences in hardwood patch patterns between 1939 and the present and between other paired groups. $\chi^2$ goodness of fit tests were used to check for differences in the distributions of historical and present-day patches across environmental gradients and ownership types. Analysis of Variance was used to test for differences between means for categorical patch metrics, followed by Fisher's Least Significant Difference test when a difference between means was present in order to determine which groups were different.

## 4. Results

### 4.1. Patch characteristics

In general hardwood patches declined in number, size and total area, and became more irregular in shape. The number of hardwood patches declined from 268 in the historical landscape to 244 in the present-day landscape. Turnover was quite high; 39% of present-day patches were new since the historical sample. The total area of hardwood cover comprising the patches declined by 40%, from 984 to 585 ha, and the mean size of patches decreased by 35%, from 3.67 to 2.40 ha (s.e. 0.47 and 0.25, respectively; $p = 0.02$). Median patch size also decreased, from 1.16 to 0.92 ha. The complexity of patch shapes increased over time, from 2.12 to 2.44 (s.e. 0.05 and 0.08; $p < 0.001$). Mean core to edge area ratios for patches declined from 0.24 to 0.10 (s.e. 0.02 and 0.01; $p < 0.0001$) as patches became smaller and more irregular in shape. The proportion of patches with within-patch cover-type heterogeneity (non-hardwood patches within hardwood patches) dropped from 86% to 26% of all patches. Within-patch heterogeneity was dominated by large mixed conifers (occurred in 41% of historical patches and 10% of present-day patches), but also included moderate amounts of very large mixed conifers (13% historically, 2% at present), medium mixed conifers (10% historically, 8% at present), and the pasture/meadow cover type (14% historically, 1% at present).

Patches that we tracked over time most often disappeared (44%), or decreased in size (41%). Median patch size of tracked patches declined from 2.04 to 1.00 ha, but those patches that persisted had a present-day median patch size that was slightly larger than the overall present-day population, at 1.08 vs. 0.92 ha. Smaller patches disappeared more often than large patches, which tended to decline in area rather than disappear. Tracked medium-sized patches were most likely to increase in size. Of those patches that persisted, smaller patches gained in area more frequently than larger ones (Table 1). The weighted mean size of patches that survived increased significantly, from 1.42 historically to 2.69 ha at present day (s.e. 0.24 and 0.43; $p < 0.005$). Surviving patches also changed in shape: 60% fragmented and increased in mean shape complexity from 2.37 to 3.29 (s.e. 0.09 and 0.15; $p < 0.0001$), and 19% coalesced and decreased in mean shape complexity from 2.57 to 1.98 (s.e. 0.13 and 0.15; $p = 0.004$). All surviving patches that changed shape decreased significantly in core to edge area ratios (Fig. 2).

Table 1
Percent of historical patches of each size class (very small to very large) and their type of change in size when historical patch area was compared to present-day

| Patch size | $n$ | Type of change in patch size | | | |
|---|---|---|---|---|---|
| | | Increased | Decreased | No change | Disappeared |
| VS (0.04-0.5 ha) | 80 | 16.3 | 13.8 | 1.3 | 68.8 |
| S (0.5–1.0 ha) | 50 | 22.0 | 28.0 | 2.0 | 48.0 |
| M (1.0–2.0 ha) | 40 | 25.0 | 35.0 | 2.5 | 37.5 |
| ML (2.0–4.0 ha) | 34 | 5.9 | 67.7 | * | 26.5 |
| L (4.0–8.0 ha) | 33 | 15.2 | 63.6 | * | 21.2 |
| VL (>8.0 ha) | 31 | 3.2 | 83.9 | * | 12.9 |
| All Patches | 268 | 14.9 | 40.7 | 1.9 | 42.5 |

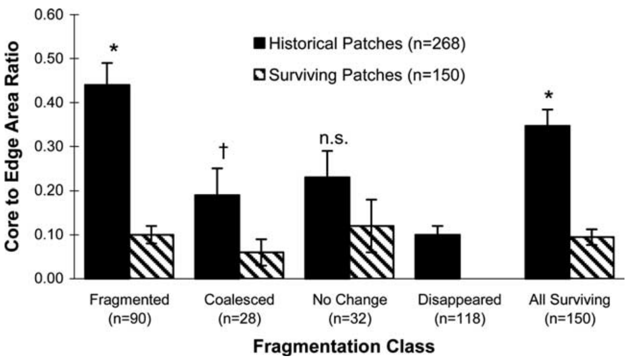

Fig. 2. Mean core to edge area ratios of historical and surviving tracked patches. Edge was the first 20 m into the patch interior from the periphery of the patch. Standard error bars are shown. Notation above bar pairs indicate $p$-values from $t$-tests: * $p < 0.0001$, † $p = 0.06$, n.s. = not significant.

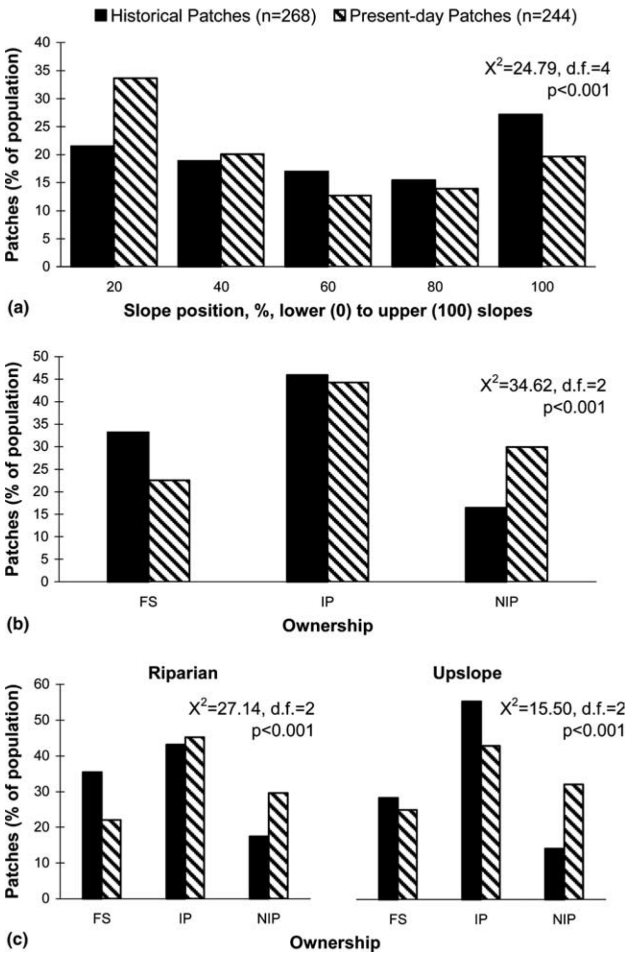

Fig. 3. Change from historical to present-day landscapes in the distribution of hardwood patches according to: (a) slope position; (b) ownership; (c) riparian and upslope location according to ownership. Ownership classes are as follows: FS = USDA Forest Service; IP = industrial private; NIP = non-industrial private. Historical riparian $n = 195$, current riparian $n = 186$. Historical upslope $n = 73$, current upslope $n = 58$. Results of $\chi^2$ tests for differences in distribution between dates are shown with each graph.

### 4.2. Patch relationships with location

The topographic distribution of hardwood patches shifted from higher toward lower slope positions ($p = 0.03$) (Fig. 3(a)). In 1939 patches were most common on ridgetops but by the end of the period they were most commonly located within the lower 20% of the slope. Distributional shifts according to stream distance ($p = 0.04$) and proportion of patch within the riparian buffer ($p = 0.04$) were very similar to those for slope position. The total hardwood area comprising the patches also followed similar trends (data not shown). Change took place in the ownership of hardwood patches, with fewer patches on Forest Service lands and more patches on non-industrial private lands at present day than had occurred there historically. Private industrial patch holdings were the highest proportion of the total (~45%, similar to their occupancy of the landscape), and remained relatively stable in number over the time period examined (Fig. 3(b)).

When we compared riparian and upland patch dynamics, we found that mean patch size decreased significantly on riparian patches ($p = 0.01$) but change in upland patch size was non-significant (Table 2). Upland patches were much smaller to start with ($p < 0.0001$) so a larger decrease in riparian patch size is not surprising. Riparian patch shape complexity was greater than that

Table 2
Change from the historical to the current landscape in riparian and upland patch mean size and shape. Column headings H*n* and C*n* are the sample sizes for the historical and current populations, respectively

| Patch location | H*n* | C*n* | Historical mean patch size (ha) | s.e. | Current mean patch size (ha) | s.e. | *p* | Historical mean patch shape | s.e. | Current mean patch shape | s.e. | *p* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Upland | 73 | 58 | 0.96 | 0.19 | 0.78 | 0.14 | n.s. | 1.65 | 0.07 | 1.83 | 0.10 | n.s. |
| Riparian | 195 | 186 | 4.69 | 0.62 | 2.90 | 0.31 | 0.01 | 2.30 | 0.06 | 2.63 | 0.10 | 0.004 |
| 1st and 2nd order stream riparian | 164 | 150 | 5.22 | 0.73 | 3.36 | 0.38 | 0.03 | 2.32 | 0.07 | 2.78 | 0.11 | 0.0005 |
| 3rd and higher order stream riparian | 85 | 92 | 6.24 | 1.23 | 4.20 | 0.56 | n.s. | 2.49 | 0.09 | 3.01 | 0.16 | 0.01 |
| Both order types riparian | 54 | 56 | 8.76 | 1.84 | 6.26 | 0.80 | n.s. | 2.68 | 0.11 | 3.64 | 0.20 | 0.0001 |
| 3rd and higher order flat riparian | 60 | 70 | 7.07 | 1.65 | 4.79 | 0.68 | n.s. | 2.60 | 0.12 | 3.22 | 0.18 | 0.005 |
| 3rd and higher order sloped riparian | 48 | 61 | 8.15 | 2.02 | 5.42 | 0.76 | n.s. | 2.71 | 0.13 | 3.33 | 0.19 | 0.01 |
| All patches | 268 | 244 | 3.67 | 0.47 | 2.40 | 0.25 | 0.02 | 2.12 | 0.05 | 2.44 | 0.08 | 0.0009 |

Sample sizes for stream orders may add to greater than the total for all riparian patches at a given date because several patches occurred along both types of stream orders (“both order types riparian”), e.g., were inclusive of a tributary junction between a lower- and a higher-order stream.

for upland patches both historically ($p < 0.0001$) and at present ($p < 0.0001$), and it increased over time. Riparian patches along smaller streams were larger and had simpler shapes historically as compared to present-day (Table 2). Along the larger streams, riparian patches that were located on flat terrain increased in shape complexity and so did patches located on sloping ground, but patch sizes at both locations did not differ over time (Table 2). Declines in the number of patches in riparian areas occurred primarily on Forest Service lands, and losses of upland patches occurred primarily on industrial private lands. Increases in number of riparian and upland patches on non-industrial private lands offset these losses somewhat (Fig. 3(c)). Patches with inholdings of large mixed conifers had a higher mean distance from streams at present day than they did historically, and those with very large mixed conifers had a lower mean distance from streams at present day than historically (Fig. 4). Historically, heterogeneous hardwood patches did not differ in mean stream distance according to cover type, but present-day hardwood patches with

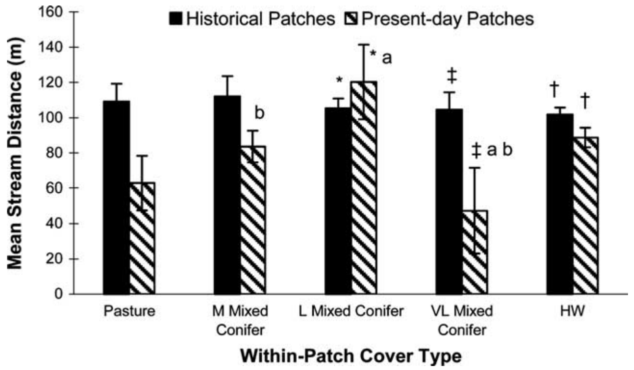

Fig. 4. Mean distance to stream of heterogeneous hardwood patches and all hardwood patches in historical and present-day landscapes. Heterogeneous patch types are separated according to the type of non-hardwood patch inclusion. Some patches had more than one non-hardwood within-patch cover type. Paired symbols or letters above standard error bars indicate *p*-values of significant differences from pairwise *t*-tests of all between-cover-type combinations within dates and within-cover-type comparisons between dates. * a, $p \leqslant 0.05$; ** $p \leqslant 0.01$; † b, $p \leqslant 0.005$; ‡ $p \leqslant 0.001$.

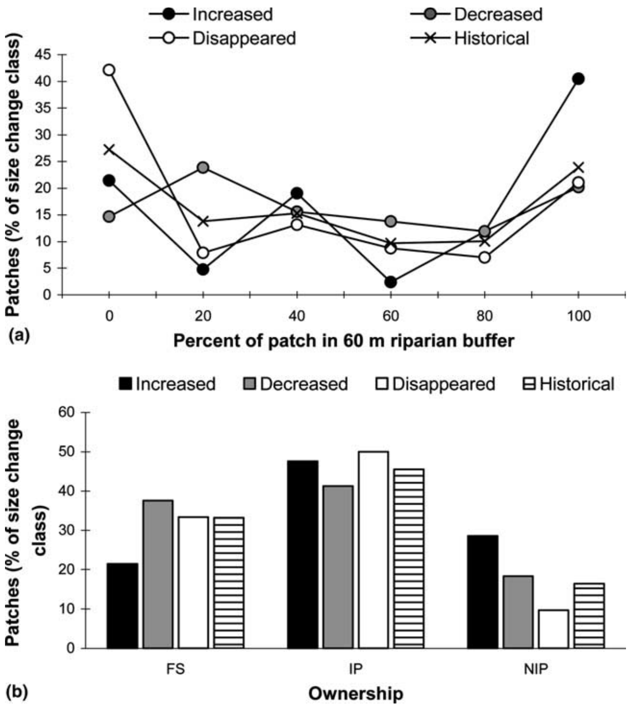

Fig. 5. Trajectories of patch size change of tracked patches in relation to: (a) percent of patch in riparian buffer, (b) ownership. Lined bar indicates the historical, initial distribution of patches as reference for change classes.

very large mixed conifer inholdings were closer to streams than those with medium or large mixed conifers (Fig. 4).

Trajectories of tracked patches indicated relationships of patch size change with stream distance, slope position, percent of patch in the 60 m riparian buffer, and ownership. Patches that increased in size tended to be located fully within the riparian buffer, and patches that disappeared tended to be located entirely without the riparian zone (Fig. 5(a)) and, relatedly, at the higher slope positions (data not shown). Tracked patches also decreased in size on the Forest Service ownership type, but increased in size on non-industrial private lands (Fig. 5(b)). Nearly 30% of patches that gained in size were located on non-industrial private lands, although only ~15% of the patches occurred on lands of this ownership class.

## 5. Discussion

### 5.1. Conservation implications

Declines in hardwood patches have numerous implications for conservation. Hardwood patches have been shown to be hotspots for native biodiversity (Neitlich and McCune, 1997). Declines in the amount of hardwoods and increasing fragmentation of hardwood patches mean less habitat is available for the mammalian, bird, insect, bryophyte and other species associated with hardwoods. For example, the yellow-billed cuckoo (*Coccyzus americanus*), a State-listed (Oregon) sensitive species, is found in dense riparian hardwood forests, the white-footed vole (*Phenacomys albipes*), a State-listed sensitive species and Federal species of concern, is most often found in riparian alder stands in coniferous forests, the western gray squirrel (*Sciurus griseus*), a State-listed sensitive species, is associated with forests having a hardwood component, and the white-tailed deer (*Odocoileus virginanus*), a State- and Federally-listed endangered species, is associated with hardwood riparian patches and oak woodlands (Csuti et al., 1997). Also, the bird species downy woodpecker (*Picoides pubescens*), acorn woodpecker (*Melanerpes formicivorus*), Wilson's warbler (*Wilsonia pusilla*), yellow warbler (*Dendroica petechia*), black-headed grosbeak (*Pheucticus melanocephalus*), warbling vireo (*Vireo gilvus*), Bullock's oriole (*Icterus bullockii*), white-breasted nuthatch (*Sitta carolinensis*), and several epiphytic lichen species, (e.g., the cyanolichen *Lobaria pulmonaria*) are among the species associated with hardwood patches (Hayes and Hagar, 2002; Neitlich and McCune, 1997). Localization of hardwoods to near-stream, lower slope position areas, where they are strong competitors (Minore and Weatherly, 1994) has the consequence of localizing to those areas any species that use hardwoods as critical habitat. Since those areas tend to be under private non-industrial land ownership, the likelihood of impact from factors associated with human settlement may be greater. Further, since this ownership class is considered the least predictable of the major ownership types in terms of future forest management and land conversion activities (Lettman, 1998), the fates of these species may be more uncertain. Losses of hardwood patches from upland areas may reduce the distribution of hardwood-associated species over time, especially for low motility species types such as epiphytic macrolichens. Additionally, because hardwoods enhance soil and stream fertility through the production of high N-content litter (Compton et al., 2003; Perry et al., 1987; Piccolo and Wipfli, 2002; Tarrant et al., 1969), reductions in hardwoods may result in declines in fertility and reduced ecosystem function. In a landscape such as the Oregon Coast Range where management focuses on promoting conifer forests and fire suppression is now practiced, managing to maintain areas of hardwood patches across the range of topographic positions and across ownerships may provide connectivity between isolated areas of habitat and thereby promote the survival and abundance of hardwood-associated species and ecosystem function.

### 5.2. Patch characteristics

The results of our study do not support the assertion that hardwoods have expanded in the central Coast Range in the last 50 years (Hibbs and Giordano, 1996; Spies et al., 2003). We found that this early successional patch type has declined and its topographic range has constricted from early logging times to the present day. In fact, hardwood patches have undergone many kinds of decline: decreased mean patch size, decreased patch shape simplicity, decreased patch core to edge area ratios, high amounts of patch fragmentation and disappearance, and restriction of patch locations toward particular topographic positions and ownerships. Our finding that patch shape is affected by management intensity supports the hypothesis of Forman (1995), and our observations of declines in size and core to edge area ratios with increased human disturbance are similar to those of Mladenoff for old-growth fragments in a northern hardwoods landscape (Mladenoff et al., 1993).

### 5.3. Patterns and sources of decline

The variation we found in hardwood patch dynamics according to ownership likely relates to variability in forest management among the ownership types. Hardwood declines on Forest Service lands are probably related to the combination of: (1) increases in conifer plantations in matrix areas, and (2) succession to conifers in forest reserves that have taken place in recent decades on Forest Service lands, as noted by Cohen et al. (2002). The location of non-industrial private lands along larger streams where growing conditions are favorable for hardwoods, and the tendency of that landowner group to use management that fosters hardwoods (i.e., selective conifer removal and less use of herbicides than industrial private landowners) (Lettman, 1998) would allow for hardwoods to increase on non-industrial private lands. There has been little shift from agriculture to forest land use in the Coast Range in the past several decades (Azuma et al., 2002), so old field succession is probably not a cause of hardwood increases on non-industrial private lands. On industrial private forest lands, we were surprised to find that hardwoods did not decrease there given the intensive management and active management against hardwoods used there since the early 1970s (Lettman and Cannon, 1998). It is possible that increases in hardwoods occurred on industrial lands early in the study period as logging increased on these lands but were then offset by declines in hardwoods later in the period as conifer regeneration was favored by managers (Cohen et al., 2002; Kennedy and Spies, in press; Wimberly and Ohmann, 2004). A limitation of our study is that our two-date sample does not capture within-period fluctuations in hardwood patch population dynamics.

The hardwood patch population was very dynamic over the study period. Forty-four percent of historical patches disappeared, and 39% of present-day patches originated during the 54-year time period covered by our study. Industrial private ownerships had the greatest rate of patch disappearance of the landowner groups, indicating that recent intensive management for conifers has occurred on these lands. Our estimate of change in hardwood forest types is more conservative than other estimates that have been made for recent decades on non-federal lands using the time series data of the forest inventory and analysis (FIA) survey collected by the USDA Forest Service (data are not yet available for federal lands). For example, between 1970s and 1980s, the probabilities of disturbance of the red alder patch type on industrial private lands and non-industrial private lands in the westside portion of the Pacific Northwest were 0.78 and 0.70, respectively, and there was a 0.28 probability that a clearcut red alder stand on industrial land would regrow as red alder, a 0.69 probability that it would become Douglas-fir, and zero probability that it would become another hardwood type (Alig et al., 2000). The time period we examined included both less intensive management than that practiced today, and the development of intensive management in the central Coast Range. We therefore would expect that the rates of hardwood patch declines we observed to be greater in the future on private lands given the documented efficacy of intensive management to preclude hardwood patch establishment and growth.

Our observations that hardwoods were increasingly restricted to riparian areas along larger streams and the limited declines of patch size in these areas relative to patch changes along smaller streams is consistent with two processes: (1) the competitive abilities of hardwoods in higher order stream riparian areas, including rapid colonization after disturbance (Harrington, 1990; Minore and Weatherly, 1994; Pabst and Spies, 2001), and (2) intensive management and conversion to conifers in uplands, in which lower-order streams are often nested. Our finding that riparian forest patches were long, narrow, and complex in shape is similar to that of Forman (1995), who characterized forest patches along streams as highly convoluted and elongated. The increase in overall shape complexity and decrease in core to edge area ratios we observed is not surprising given that patches became more restricted over time to these near-stream areas. Cover-type simplification within hardwood patches was also probably related somewhat to the shift of the patch population toward near-stream areas where hardwoods are highly competitive and where selective removal of conifers from riparian buffers has been allowed (Oregon Department of Forestry, 2001).

The current pattern of hardwoods probably results from a combination of the topographic positions, vegetation types and management practices that vary by ownership, and from the historical conditions particular to the study area. Expansion of hardwoods beyond near-stream areas may have occurred prior to the 1940s, following the large fires that occurred in the settlement period. Hardwoods may be declining across the landscape and becoming restricted to near-stream areas partly in relation to this prior expansion. Hardwoods may also be declining and being increasingly restricted to particular locales because of the combination of lower intensity management on non-industrial private lands relative to the other ownerships, and the location of non-industrial private lands along larger streams (Azuma et al., 2002). Given that the historical range of variability of hardwood patches is unknown, it is uncertain whether hardwoods are declining from higher-than-average amounts or how potential future amounts of hardwoods may relate to historical levels. Nonetheless, the palynological record indicates that hardwoods, in particular red alder and *Quercus* spp., have been present in the central part of the Oregon Coast Range at moderate to high levels throughout the Holocene (Long et al., 1998).

The present-day practice of establishing riparian buffers along perennial streams to protect fish habitat may provide refugia for hardwoods in these intensively managed landscapes in this region where climatic conditions favor conifers (Waring and Franklin, 1979). On the other hand, current forest management for large conifers near streams (Oregon Department of Forestry,

2001) to provide shade and in-stream structure for salmon habitat may reduce hardwoods in some places where they are competitive.

In other areas of the Pacific Northwest and in other coniferous regions such as Spruce-fir forests of the northeastern US, and boreal forests of Scandinavia (Essen et al., 1997), we would expect that, where conifers have supplanted hardwoods through management practices and fire frequencies and timing are similar to those of the study area, hardwoods would probably be in decline from early-logging conditions if other compensating factors allowing for hardwood expansion are not present. In forests where the fire frequency was historically low, small gap disturbance is more prevalent, and late successional characteristics may predominate, such as some areas of coastal British Columbia (Lertzman et al., 1996), early successional hardwoods might be expected to increase with clearcut harvest, depending on the reforestation methods used, site characteristics, and the relative efficacy of conifer establishment. It also quite possible that other forest regions that experienced widespread logging or fires in the late 19th century and early 20th century developed large patches of early successional hardwoods that are now being reduced by conversion to conifer plantations on private lands and successional losses to conifers on lands managed for late-successional conifer forest habitats. In general, at the landscape level, where hardwoods are found as an early successional vegetation type and succession has advanced and/or non-hardwood types have been promoted through management in recent decades, hardwoods are likely to be in decline. Although human disturbance in the past in conifer forest landscapes may have led to increases in abundance of hardwoods, human activities in the future may greatly reduce the distribution of hardwoods and associated plant and animal species.

### *5.4. Characterizing patch patterns*

Hardwoods in this forest system are a minor patch type that at least superficially fits a 'Swiss cheese' model of landscape dynamics in which small disturbances occur within a dominant matrix and then fill in with different types of vegetation (Pastor and Broschart, 1990). Yet, as Lieberman et al. (1989) point out this simple model, which has been applied to canopy gap dynamics, does not do justice to the complexity of forests. For example, we observed that the 'holes' (i.e., hardwood patches) themselves had holes (i.e., conifer or other cover-type patches within the hardwood patches), and that patch boundaries and within-patch cover types were complex and highly variable over time and space.

Few other studies have considered nested patches or within-patch cover type heterogeneity. The perforation of patches by other patch types, or the nesting of patches, has been described generally as a variant of unperforated patches (Forman, 1995). Our finding that nested patches were quite common in the historical landscape but scarce at present indicates the importance of considering nested patches and monitoring change in within-patch heterogeneity. The loss we found of large conifers within hardwood patches is consistent with documented declines in large conifers across the entire Coast Range (Cohen et al., 2002; Wimberly and Ohmann, 2004), and also signifies a loss of intermixing of patches. However, it is possible that any shift in ecological function that results when small groups of large conifers are removed from a hardwood patch differs from changes that take place when they are removed from a coniferous matrix.

### *5.5. Conclusions*

We found that both human activities and environmental factors have influenced the dynamics of hardwood patches from post-settlement historical times to the present day. The conditions we found in the central Oregon Coast Range may be one example of a more widespread phenomenon resulting from the conversion of hardwood/mixed conifer forests to managed conifer stands. Human disturbances may either increase or decrease landscape complexity depending on the scale of evaluation, the attribute examined, and the intensity of human activity (Forman, 1995; Mladenoff et al., 1993). In this study, which covered the transition from early logging to more intensive logging, increasing disturbance from forest management activities resulted in a landscape in which early-to-mid successional hardwood patch sizes were smaller and patch shapes more convoluted. Thus, the generality that increasing human influence necessarily creates larger early successional patch types and simpler shapes does not necessarily apply. In fact, with human influence, the landscape representation of some early successional patch types such as hardwoods may be reduced or restricted to particular locales and become fragmented just as later successional types do. This may result in a decline of habitat for species that are associated with early successional patch types.

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