

Extent and spatial patterns of grass bald land cover change (1948–2000), Oregon Coast Range, USA

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Abstract Globally, temperate grasslands and meadows have sharply declined in spatial extent. Loss and fragmentation of grasslands and meadows may impact biodiversity, carbon storage, energy balance, and climate change. In the Pacific Northwest region of North America, grasslands and meadows have declined in extent over the past century. Largely undocumented in this regional decline are the grass balds of the Oregon Coast Range, isolated grasslands in a landscape dominated by coniferous forests. This study was conducted to quantify the spatial extent and patterns of grass bald change. Five balds in the Oregon Coast Range were evaluated using historical aerial photographs and recent digital orthophoto quadrangles (DOQ). Over the time period of study (1948/1953 to 1994/2000), bald area declined by 66%, primarily from forest encroachment. The number and average size of bald vegetation patches declined, while edge density increased. Tree encroachment into balds was inversely related to distance from nearest potential parent trees. Spatial patterns of bald loss may result from a forest to bald gradient of unfavorable environmental conditions for tree establishment and/or seed dispersal limitation. Species dependent on balds may be at risk from loss of bald area and increased fragmentation, although metrics of habitat fragmentation may not reflect species-

specific habitat requirements. Tree encroachment patterns and increased bald edge densities suggest increasing rates of bald loss in the future. The remote sensing nature of this study cannot determine the fundamental causes of bald decline, although prior research suggests climate change, cessation of native burning, successional changes in response to prior wildfires, and cessation of livestock grazing all may have potential influence.

Keywords Grassland decline · Land cover change · Forest encroachment · Habitat loss · Fragmentation

Introduction

Declines in the extent of temperate grasslands, montane meadows, and subalpine meadows appear to be a global ecological phenomenon (Mast et al. 1997; Bowman et al. 2001; Didier 2001; League and Veblen 2006; Marie-Pierre et al. 2006; Baker and Moseley 2007; Coop and Givnish 2007). Noting that conversions of temperate grasslands to other land cover types greatly exceeds conservation; Hoesktra et al. (2005) consider the loss of grasslands as a major component of a global “biome crisis”. Extensive losses of grassland habitat will reduce biodiversity, since habitat modification and fragmentation are believed to be significant threats to biodiversity (Sala et al. 2000;

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Foley et al. 2005). Grasslands and meadows occupy a significant portion of global land surface, and are major contributors to biosphere primary production (Field et al. 1998; Saugier et al. 2001). Large-scale declines of grasslands and meadows may change regional carbon storage (Houghton et al. 1999; Jackson et al. 2002; Knapp et al. 2008), and surface albedo (Bonan et al. 1992), thereby influencing global climate change.

In the Pacific Northwest Region of North America, the loss of lowland, montane, and subalpine grasslands have been largely attributed to tree encroachment (Habeck 1961; Franklin et al. 1971; Woodward et al. 1995; Rochefort and Peterson 1996; Miller and Halpern 1998; Foster and Shaff 2003), although agricultural development has played a major role in the loss of Willamette Valley prairies (Habeck 1961; Johannessen et al. 1971). Largely overlooked within the region have been the grass balds of the Oregon Coast Range, graminoid dominated vegetation which typically occur at the summits of Coast range peaks and contrast strikingly in structure with the taller surrounding coniferous forests (Franklin and Dyrness 1998). Like balds of the Southern Appalachian Mountains of North America, these balds had been relevantly stable in extent, while containing rare and endemic plant species, and disjunct populations, suggesting they are relict vegetation assemblages (Merkle 1951; Detling 1953; Wiegand and Knowles 1995). However, there is considerable uncertainty regarding the origins, maintenance, and decline of the balds, or the role of humans in bald dynamics. Coast Range balds provide habitat to multiple species with varying levels of Federal and State protected status (USFWS 2001; ONHIC 2007). Their limited spatial extent, unique vegetation, and presence of disjunct and rare species contribute to their ecological importance and value to society.

Limited research within the balds has focused on vegetation classification (Merkle 1951; Aldrich 1972; Snow 1984). Magee and Antos (1992) investigated tree encroachment into one Coast Range bald, but no published documentation exists quantifying current and historical bald extent, or changes in bald area over time. There is no documentation of the types of bald loss (i.e., forest encroachment versus human disturbance), spatial patterns of bald loss, or landscape metrics (i.e., number of bald patches, area of patches, and edge density of patches) of bald vegetation fragmentation over time. Quantifying spatial patterns

of bald change may provide information to make inferences about the underlying proximate processes involved. Landscape metrics of fragmentation are a widely used proxy for habitat degradation. Mapping of bald change, and the extent of bald fragmentation could provide land managers with useful spatially explicit information for establishing reference conditions and prioritizing specific areas for restoration activities.

The objectives of this study were to: (1) quantify the extent of balds in the Oregon Coast Range over the past five decades (1948–2000), (2) determine what types of land cover balds are being converted to (i.e., coniferous forest, roads and building, bare ground, etc.), (3) quantify the spatial patterns of tree encroachment into the balds, and (4) determine if bald vegetation patches have become more fragmented and edgier (i.e., decline in total number of patches and average patch size, while having increased edge densities) over the time period of study.

Methods

Study area

Franklin and Dyrness (1998) listed seven major balds in the Oregon Coast Range, but omitted the extensive bald on Mount Hebo (USFWS 2001). Of these eight balds, only five were used in this study due to the poor quality of historical imagery at three balds (Fig. 1). From north to south, balds studied are: Mount Hebo, Bald Mountain, Marys Peak, Grass Mountain, and Prairie Peak (see Table 1 for study area information). Bald elevations range from 968 to 1249 m. All five balds are in the *Tsuga heterophylla* zone, Coast Range Level III Ecoregion (Franklin and Dyrness 1998; Thorson et al. 2003). However, higher balds (such as Marys Peak) may lie within the higher elevation, *Abies amabilis* zone (McCain and Diaz 2002). The climate is maritime, with mild wet winters and cool dry summers, but climate varies with proximity to the ocean, latitude, and orographic effects. Over the 1970–2000 time period, climatic data from the nearest weather station (Laurel Mountain Weather Station, National Weather Service Cooperative Network #354776, 44°55' N 123°34' W, 1090 m), reported annual average maximum temperature of 10°C, mean temperature of 6.6°C, annual average minimum temperatures of 3.1°C, annual



Fig. 1 Documented major grass balds within the Oregon Coast Range ecoregion. Grass Balds names are: SM—Saddle Mountain, MH—Mount Hebo, BM—Bald Mountain, MP—Marys Peak, GM—Grass Mountain, PP—Prairie Peak, RN—Roman Nose Mountain, and TY—Tyee Mountain. Grass Balds with solid triangles were studied

precipitation of 307 cm, and total snowfall of 301.5 cm which falls predominantly from November to April. Mount Hebo, Bald Mountain, and Prairie Peak study areas are predominantly underlain by mafic intrusions of Oligocene origin, while Marys Peak and Grass Mountain study areas are underlain by a combination of Oligocene mafic intrusions and middle Eocene sandstones and siltstones (Walker and MacLeod 1991).

Bald vegetation varied between study areas. Common graminoid species include: *Elymus glaucus*, *Festuca rubra*, *Bromus carinatus*, *Poa pratensis* (non-native), *Agrostis diegoensis*, *Carex californica*, and *Carex rossii*. Common forbs and ferns include: *Lomatium martendalei*, *Collinsia parviflora*, *Senecio triangularis*, *Delphinium menziesii*, *Iris tenax*, and *Pteridium aquilinum* (see, Merkle 1951; Aldrich 1972; Schuller and Exeter 2007 for details). Disjunct populations of more typically Eastern Cascades species include *Festuca idahoensis* and *Carex hoodii*, while populations of species more typical of high altitudes include *Phleum alpinum*, *Arctostaphylos uva-ursi*, *Phlox cepitosa*, *Silene douglasii*, *Allium crenulatum*, and *Lupinus lepidus* var. *lobbi* (Merkle 1951; Detling 1953; Oregon Plant Atlas 2007). Multiple species of concern are associated with grass bald vegetation in the Oregon Coast Range, including: the Federally listed Oregon Silverspot Butterfly (*Speyeria zerene hippolyta*), the State listed Coast Range Fawn Lilly (*Erythronium elegans*), and three plants that are candidates for State protection (*Cardamine pattersonii*, *Saxifraga hitchcockiana*, and *Sidalcea hirtipes*) (USFWS 2001; ONHIC 2007). Nomenclature of plant species follows Hitchcock and Cronquist (1976). Multiple conifer species can encroach into the balds. At Mount Hebo *Psuedotsuga menziesii* dominates encroachment (personal observation). *P. menziesii* is also the dominant encroaching species at Bald Mountain and Prairie Peak. *P. menziesii*, *Abies procera*, and to a lesser degree *Tsuga heterophylla* are encroaching at Grass Mountain, while *A. procera* is the dominant encroaching species at Marys Peak (Magee and Antos 1992; Schuller and Exeter 2007).

Aerial photo change detection

Analyses used historical (1948 and 1953) aerial photographs and recent (years 1994 and 2000) digital orthophoto quadrangles (DOQ's) of the five study areas. The historical images were geo-rectified using the DOQ's as reference imagery. Historical photos and DOQ's were filtered to improve classification of imagery into distinct land cover classes. Filtered images were manually classified into four land cover classes (bald, coniferous forest, roads and buildings, and bare ground) within each image. A matrix function was applied to these classified images, resulting in a single image with 16 (4 × 4) potential land cover

Table 1 Grass bald study area maximum elevations, geographic coordinates, and imagery dates

Study area	Elevation (m)	Latitude	Longitude	Imagery dates (dd/mm/year)
Mount Hebo	968	45°12' N	123°45' W	29/07/1953, 30/05/1994
Bald Mountain	985	44°47' N	123°32' W	29/06/1948, 08/08/2000
Marys Peak	1,249	44°30' N	123°33' W	14/07/1948, 30/08/1994
Grass Mountain	1,098	44°27' N	123°33' W	29/06/1948, 30/05/1994
Prairie Peak	1,043	44°16' N	123°37' W	29/06/1948 ^a , 02/07/1948 ^a , 30/08/1994

^a Two historical images were mosaicked for Prairie Peak. See Appendix A for details

change classes for each study area. Many of these 16 change classes were irrelevant for this study (such as coniferous forests converted to roads and buildings), so the numbers of land cover classifications were reduced to aid in visual interpretation of bald change (Fig. 2). The surface area of the remaining relevant cover classes were compiled to determine the amount of historical and current bald area, the extent of bald change, and what types of land cover balds were being converted to (i.e., coniferous forest, roads and buildings, and bare ground). Unless otherwise mentioned, image processing and analyses took place using ERDAS Imagine 8.7 (2003 Leica Geosystems GIS and Mapping, LLC). For a detailed description of image acquisition and processing methods please refer to Appendix A.

Spatial pattern metrics

Landscape metrics were calculated to quantify bald patch fragmentation. The number of bald patches, average patch size (referred to as patch size), area-weighted average patch size (referred to as weighted patch size), and edge density were calculated for both historical and recent classified images. Weighted patch size was calculated in addition to simple patch size because the distribution of bald patch sizes was skewed to a few large patches and many small patches. In this patch distribution, the simple arithmetic average will not reflect the expected patch size encountered by random placement of points, resulting in a disproportionate effect of small patches (Turner et al. 2001). The weighted patch size was calculated as,

$$S_a = \frac{\sum (S_k^2)}{\sum (S_k)},$$

where S_a is the area-weighted average patch size, and S_k is the size of the k th patch. Edge density was calculated as the sum of the lengths (m) of all edge segments within the bald patches divided by the

landscape area (m²) of all bald patches, multiplied by 10,000 (to convert to meters of edge per hectare). Higher edge density values are indicative of complex or elongated patches, lower edge density values indicate more compact or simple shaped patches.

The function Distance in the Spatial Analysis of ArcGIS 9.1 (2005 ERSI Inc.) was used to test the relationship between tree encroachment and distance to potential parent trees. The Euclidean distance was calculated between each pixel of bald encroached by trees and the nearest pixel of potential parent tree (i.e., nearest forested pixel in the historical imagery). Euclidean distance was also calculated between each pixel of bald that remained stable (did not change between imagery dates) and the nearest pixel of potential parent tree. The percentage of pixels within both encroached bald and unchanged bald classes were separately rescaled to total 100%, and plotted with respect to distance from nearest potential parent trees. A Wilcoxon signed rank sum test was used to determine whether distribution of pixels with respect to distance from potential parent trees differed between encroached bald and unchanged bald pixels. Distributions were considered different if the Wilcoxon signed rank sum statistic (S value) was significant at $\alpha = 0.05$ level.

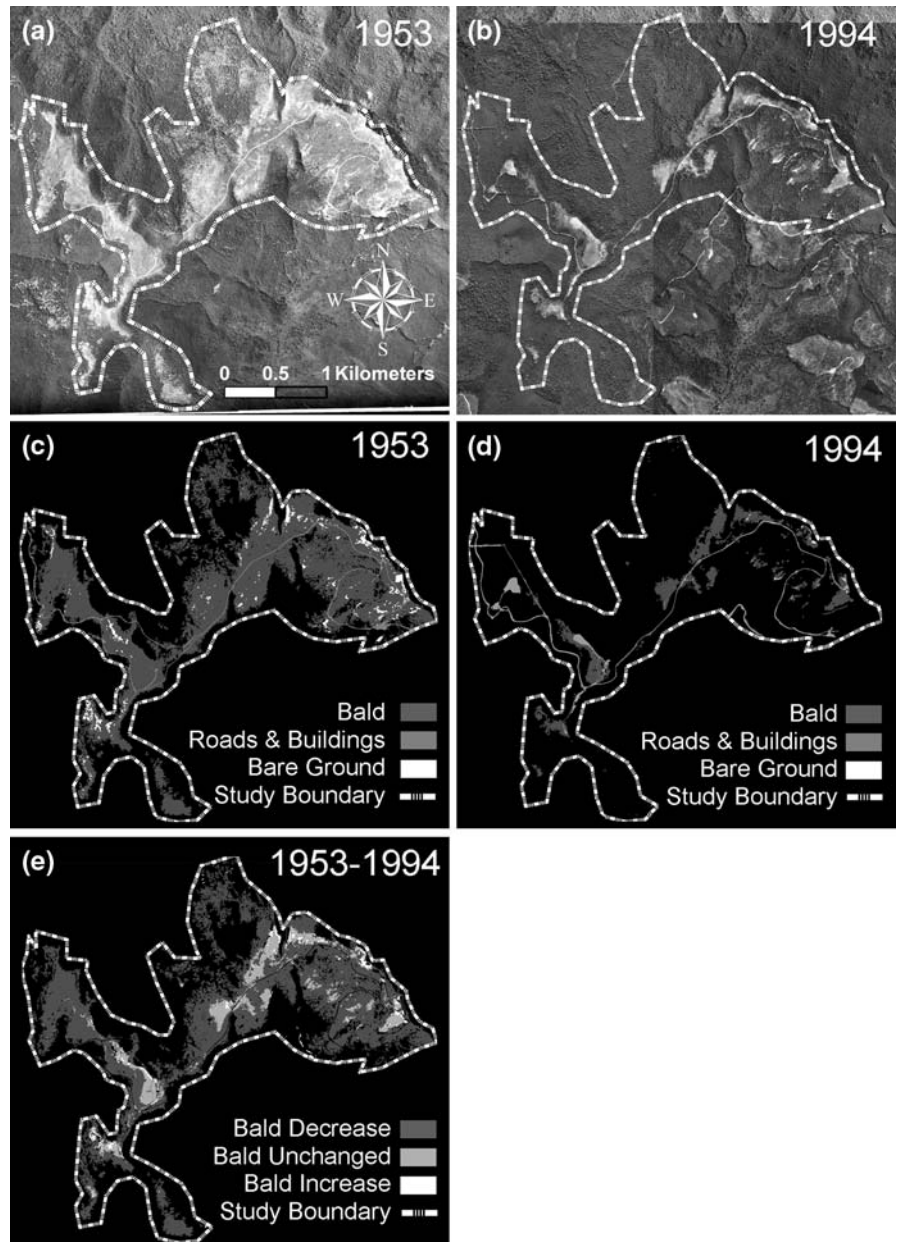
Results

Extent and decline of grass balds

All five balds declined in area over the time period of study (Table 2). From 1948/1953 to 1994/2000, balds declined from 523.6 ha to 179.2 ha (66%). Forest encroachment into balds was the dominant type of bald decline (348.7 ha, 95%). Conversion of balds to roads and buildings accounted for 15.4 ha (4%), while conversion to bare ground was minor.

Mount Hebo displayed the greatest bald decline. Bald Mountain was the smallest historical bald, had

Fig. 2 Example (Mount Hebo study area) of aerial photo grass bald change detection process. **a** 1953 historical aerial photo, **b** 1994 DOQ, **c** 1953 classified image, **d** 1994 classified image, and **e** matrix image depicting grass bald change at Mount Hebo from 1953 to 1994



the lowest proportional bald decline, and the greatest proportional amount of bald area decline due to road and building development. Visual interpretation of the Bald Mountain imagery (not shown) suggests the 0.2 ha conversion of bald to bare ground resulted from adjacent logging operations in the southern edge of the study area. Prairie Peak had the second highest percentage of bald decline. Bald decline at Prairie Peak resulted in the near complete elimination of one bald in the eastern part of the study area, while

converting the contiguous main bald into a series of smaller interconnected balds (not shown).

Spatial patterns of grass bald change

Landscape metrics of bald fragmentation (number of bald patches, patch size, weighted patch size, and edge density) were generally consistent between study areas (Table 3). From 1948/1953 to 1994/2000, the number of bald patches declined at all study

Table 2 Grass bald changes by area and types of land cover change

Study area	Date	Area (ha)	Type change ^a	Δ (ha) ^b	Δ (%) ^c
Mount Hebo	1953	243.4	Bald to forest	202.1	96.09
	1994	39.8	Bald to bare ground	0.2	0.10
	1953–1994	–203.6	Bald to R&B	8.0	3.79
			Conversion to bald	6.6	
Bald Mountain	1948	24.3	Bald to forest	14.8	84.8
	2000	7.3	Bald to bare ground	0.2	1.2
	1948–2000	–17.1	Bald to R&B	2.4	14.0
			Conversion to bald	0.4	
Marys Peak	1948	118.4	Bald to forest	47.0	93.9
	1994	77.2	Bald to bare ground	0.4	0.7
	1948–1994	–41.1	Bald to R&B	2.7	5.5
			Conversion to bald	9.0	
Grass Mountain	1948	43.9	Bald to forest	22.0	98.3
	1994	23.1	Bald to bare ground	0.0	0.0
	1948–1994	–20.8	Bald to R&B	0.4	1.9
			Conversion to bald	1.6	
Prairie Peak	1948	93.6	Bald to forest	62.7	95.4
	1994	31.8	Bald to bare ground	1.1	1.7
	1948–1994	–61.8	Bald to R&B	1.9	2.8
			Conversion to bald	3.9	
All Study Areas	1948/1953	523.6	Bald to forest	348.7	95.3
	1994/2000	179.2	Bald to bare ground	1.9	0.5
	Change	–344.4	Bald to R&B	15.4	4.2
			Conversion to bald	21.6	–

^a Type change refers to what type of land cover balds are converted to. R&B refers to roads and buildings

^b Δ (ha) refers to the area (in hectares) of each type of land cover change

^c Δ (%) refers to the proportion of bald loss due to a specific type of land cover change

Note: Bald area values used to calculate Δ (ha) and Δ (%) are bald loss values corrected for conversions of non-bald land cover classes to bald vegetation

areas (47% total 10–67% for individual study areas). Patch size for four out of the five balds declined (49% average, 20–66% for individual study areas). The fifth study area (Marys Peak) saw a 2% increase in patch size. Weighted patch size declined sharply in all study areas (94% average, 48–97% for individual study areas). Bald edge density increased at four out of the five study areas (18% total, 7–46% for individual study areas). The fifth study area (Marys Peak) saw only a 0.41% increase in edge density.

Forest encroachment into the balds was inversely related to the distance from potential parent trees (Fig. 3). The distribution of encroached bald versus unchanged bald in relation to distance from potential parent trees was significant at Mount Hebo ($S =$

-2184 , $P < 0.0001$), Marys Peak ($S = -8333.5$, $P < 0.0001$), Grass Mountain ($S = -1557$, $P < 0.0001$), and Prairie Peak ($S = -908.5$, $P < 0.0001$). Results for Bald Mountain ($S = -876.5$, $P = 0.0836$) suggest a similar relationship, but also displayed longer distance tree encroachment.

Discussion

Grass bald decline and fragmentation

This study is the first to document the widespread decline and fragmentation of grass balds in the Oregon Coast Range. All five study areas displayed

Table 3 Landscape metrics of grass balds by study area and date

Study area	Date	No. of bald patches (m ²)	Patch size (m ²)	Weighted patch size (m ²)	Edge density (m ² /ha)
Bald Mountain	1948	87	2,945	226,910	15,021
	2000	78	1,352	24,893	19,354
Grass Mountain	1948	778	595	66,473	14,957
	1994	515	473	34,398	15,983
Marys Peak	1948	1,651	791	597,727	14,660
	1994	1,066	809	202,988	14,719
Mount Hebo	1953	2,446	1,119	2199,737	13,911
	1994	1,324	380	63,454	16,081
Prairie Peak	1948	2,025	724	788,861	9,563
	1994	674	538	64,133	13,996
All study areas	1948/1953	6,987	5,384	3281,981	68,111
	1994/2000	3,657	2,744	186,878	80,134

See section “Methods” for detailed descriptions of landscape metrics

large losses in bald area due to tree encroachment, with only 35% of 1948/1953 bald area present in 1994/2000. Study areas displayed declining numbers of bald patches, reduced patch sizes and weighted patch sizes, and increased edge densities. However, these metrics of fragmentation varied by study area, suggesting different patterns of bald change. At Bald Mountain there were small declines in the number of patches, large declines of patch size and weighted patch size, and large increases in edge density. This is consistent with the elimination of only a few small bald patches, but larger declines in the area of larger patches, and edgier (i.e., more complex and/or elongated) remnant balds. Mount Hebo and Prairie Peak had the greatest declines in the number of bald patches, patch size, and weighted patch size, but had different changes in edge density. Prairie Peak bald patches had a much greater increase in edge density compared to Mount Hebo, suggesting that while both of these study areas experience declines in both large and small bald patches, the remnant patches are Prairie Peak have more elongated or complex shapes than those at Mount Hebo. In contrast, Marys Peak had a large reduction in the number of bald patches, a slight increase in patch size, a large reduction in weighted patch size, and almost no increase in edge density. These changes are consistent with the predominant disappearance of smaller patches, and reductions of remaining large patches areas while maintaining less complex and/or elongated shapes. Grass Mountain had a small increase in edge density, and when combined with the other patch metrics,

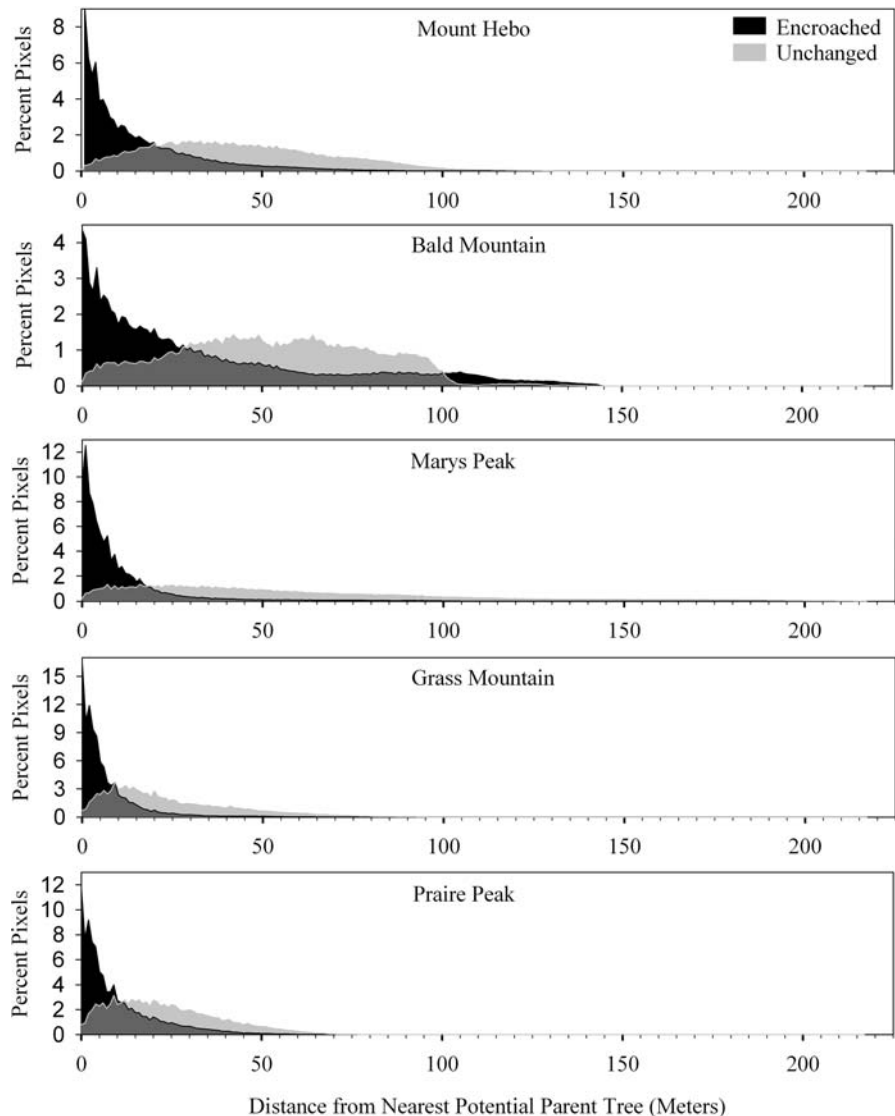
suggests bald decline was largely due to the elimination of small patches, and the shrinking of larger patches without becoming very complex or elongated in shape.

Spatial patterns and proximate causes of tree encroachment

Bald decline was overwhelming due to tree encroachment, which was greatly closer to the forest edge. This infilling pattern of encroachment is consistent with field data from Marys Peak (Magee and Antos 1992), but differs from more clustered patterns of encroachment found in montane and subalpine meadows of the Oregon and Washington Cascades (Franklin et al. 1971; Miller and Halpern 1998).

The spatial patterns of tree encroachment likely result from a combination of unfavorable tree establishment conditions due to high resource competition from graminoids, and dispersal limitation of tree seed with increased distance from the forest edge. The influence of forest edge on non-forest microsite conditions decline with increased distance from the forest edge (Chen et al. 1993). With increased distance from the forest edge there is increased graminoid cover. For many conifer species, both establishment and growth decline with increased graminoid cover (Alan and Lee 1989; Kolb and Robberecht 1996; Dovciak et al. 2008). *A. procera* germinants and seedlings tend to be at a competitive disadvantage against dense bald vegetation (Magee and Antos 1992). *Pseudotsuga menziesii* and *Tsuga heterophylla*

Fig. 3 Distribution of encroached versus unchanged grass bald land cover in relation to distance from potential parent trees



establishment and growth also decline with increases in herbaceous vegetation competition (Rose et al. 1999; Rose and Ketchum 2002). Wind-dispersed seeds can travel quite far (Clark et al. 1999), but seed dispersal displays inverse seed density with increased distance from source trees. The heavy weight of *A. procera* seeds results in a relatively short dispersal distances compared to *P. menziesii* and *T. heterophylla*. However, almost all bald vegetation was within 100 m of pre-existing trees, well within dispersal distances of invading species (Isaac 1930; Franklin and Smith 1974; Carkin et al 1978).

In the Carpathian Mountains, Dovciak et al. (2008) found invasion of grasslands by *Picea abies* to be

driven by a multivariant gradient from the forest to core grassland of seed dispersal and environmental conditions (overstory cover, moss and grass cover, etc.). Although specific microsite information is lacking for this study, prior studies within the grass balds and in other grasslands suggest both seed dispersal and environmental conditions appear to drive spatial patterns of forest encroachment.

Ecological implications

Multiple species of concern are associated with bald vegetation in the Oregon Coast Range (USFWS 2001; ONHIC 2007). Declines in habitat area and increased

fragmentation are generally associated with the loss of species diversity and reductions in species populations (Saunders et al. 1991; Sala et al. 2000; Krauss et al. 2004; Foley et al. 2005). Although, landscape metrics, such as habitat area and fragmentation can be important, they cannot directly quantify habitat quality or species composition. Reliance on landscape metrics is often necessary in the absence of, but may be less important than, species-specific field-based measures of habitat suitability (Hobbs and Yates 2003; Fischer and Lindenmayer 2007). Additionally, reductions in species populations and species richness may lag temporally behind habitat loss, further complicating the relationship between species and habitat (Tillman et al. 2002; Helm et al. 2006). Despite these uncertainties, large declines in bald area and increased bald fragmentation are likely to negatively impact rare species and disjunct populations.

Patterns of tree encroachment and bald fragmentation suggest a positive feedback of accelerating encroachment in the future. In some meadow types, tree establishment positively alters microsite conditions with respect to future tree regeneration and establishment (Miller and Halpern 1998; Dovčiak et al. 2008). Since the remnant bald patches are closer to forest edge, they are more likely to encounter a higher density of tree seed. Higher edge density of remnant bald patches also suggests the surrounding forest matrix will have increased influence on ecological patterns and processes in the remnant patches (Saunders et al. 1991). In particular, increased shade from the forest edge may reduce graminoid cover that excludes conifer establishment. If conservation and restoration of balds is desired, the potential for additional and accelerating bald loss should be a management concern. Bald decline due to road and building development was minor at most study areas (except the Bald Mountain study area). However, landscape metrics used in this study cannot quantify changes in habitat quality that occur even if the land cover type remains unchanged, while roads can increase the dispersal and establishment of non-native plant species (Parendes and Jones 2000; Trombulak and Frissell 2000; Christen and Matlack 2006).

Fundamental causes of bald decline

The proximate cause of bald decline is tree encroachment, but the primary objectives and data of this

study cannot provide empirical evidence regarding its fundamental causes. However, a review of regional causes of tree encroachment, combined with site specific disturbance and land use history, may provide valuable information. Regional causes of tree encroachment vary by study, but most often include: climate change, fire and fire suppression, grazing history, and succession.

Climate changes (via changes in growing season length, snow free periods, growing season temperatures, etc.) are a likely cause of regional subalpine tree encroachment (Franklin et al. 1971; Woodward et al. 1995; Rochefort and Peterson 1996). Tree establishment in the balds of Marys Peak coincided with favorable climatic conditions (Magee and Antos 1992). However, as previously noted, Marys Peak is at a higher elevation and contains a more subalpine forest type than other studied balds, so conifer establishment–climate relationships at Marys Peak may not be representative of other lower elevation balds. Livestock grazing also can be an important driver of tree encroachment, reducing plant cover and density, while exposing bare soil. Miller and Halpern (1998) concluded that tree encroachment in some upland meadows in the Oregon Cascades occurred following grazing cessation and coincidentally, favorable climatic conditions. Homesteaders grazed livestock on the balds from the 1800s to 1940s (Reynolds 1993), so observed forest encroachment could be partly due to grazing cessation prior to historical imagery.

High-severity wildfires are an important disturbance agent in Coast Range forests that has varied considerably over time. About 4600 to 2700 years BP, the Oregon Coast Range used to have a 140 ± 30 year fire intervals, compared to 240 ± 30 year from 2700 years ago to present (Long and Whitlock 2002). Native Americans are believed to have used frequent low-severity fire to manage some of the Coast Range grass balds prior to Euro-American homesteading (personal communication Phyllis Steeves, Archeologist, Siuslaw National Forest). By 1860, disease-driven declines in Native populations likely resulted in dramatic reductions of Native fire ignitions on the grass balds. Elimination of Native American ignitions, combined with active fire suppression, has driven tree encroachment in the prairies of the Puget Sound and Willamette Valley (Johannessen et al. 1971; Foster and Shaff 2003;

Boyd 1999). Evidence of extensive fires in the region could suggest grass balds are early-successional vegetation that has since been invaded after over 50 years of fire suppression. However, there is strong evidence balds have been long term features on the landscape: balds generally lack biological legacies (i.e., large woody debris, stumps, snags, and remnant trees) present even after multiple disturbances, they are dominated by graminoids rather than shrub-dominated early-successional vegetation typical in the Coast Range, and contain disjunct and endemic plant species suggestive of long term vegetation stability (Merkle 1951; Detling 1953; Dyrness 1973; Hemstrom and Logan 1986; Franklin et al. 2002). One potential exception is Mount Hebo, where high intensity fires, active reforestation, commissioning and decommission of the Mount Hebo Air Station, along with mowing and burning to maintain habitat for the Oregon Silverspot Butterfly, create a very complex disturbance history (Munger 1944; Oregon Division of Forestry 2001; USFWS 2001). Climate change and complex disturbance history likely interact to shape spatial and temporal patterns of tree invasion into the grass balds, as found elsewhere (Miller and Halpern 1998). However, addressing these fundamental drivers of tree invasion is beyond the scope of this study. Spatially-explicit tree age and growth chronology data, combined with climate records and detailed grazing history, may allow inference regarding the fundamental causes of tree invasion into these unique landscape features.

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Appendix A. Aerial photograph acquisition and image processing

Aerial photo acquisition

Historical panchromatic 1:12,000 scale aerial photographs were acquired from the University of Oregon

Map Library in Eugene, OR. All historical aerial photographs were taken during the summer of 1948 (except Mount Hebo, taken during the summer of 1953). Historical aerial photographs were scanned on a flatbed scanner at 600 dpi. Scanned historical images had 8-bit radiometric resolution and 0.63–0.89 m postrectification spatial resolution, re-sampled to 1 m pixel size after geo-rectification using nearest neighbor interpolation. Recent images of the study areas are panchromatic digital orthophoto quadrangles (DOQs). DOQs were downloaded from TerraServer-USA (<http://terraserver-usa.com/default.aspx>). The Mount Hebo, Marys Peak, Grass Mountain, and Prairie Peak DOQs were taken in the summer of 1994 (except Bald Mountain, taken during the summer of 2000). DOQs were in Universal Transverse Mercator Projection (UTM, Zone 10N), using the Geodetic Reference System Spheroid of 1980, and the North American Datum of 1983. All DOQs had 8-bit radiometric resolution and 1 m spatial resolution.

Aerial photo geo-rectification

Each scanned historical image was first manually clipped so only grass balds and surrounding forest vegetation were retained. Each clipped image was geo-rectified using the recent DOQ of the same site as the reference image. Control Points (CPs) between historic and recent images were manually selected, focusing on the most temporally stable features (i.e., buildings, roads, rock outcrops, and individual open grown trees). A minimum of 20 CPs between historical and reference imagery was desired, although this was not possible in all cases. Geo-rectification models used polynomial equations, evaluated by the Root Mean Squared Error (RMSE), and rejected if the RMSE exceeded 0.5 pixels. Additionally, rectified images were digitally overlain as partial transparencies over recent DOQs to qualitatively assess rectification accuracy. Rectification and resampling occurred using the nearest neighbor method, conserving raw image brightness values. A mosaic of two rectified images was required for the Prairie Peak study area (Prairie Peak a & b), and was generated using histogram matching, no cutlines, minimum select function, and union of all inputs.

Study area	Date	CPs	RMSE X	RMSE Y	RMSE total	Polynomial order
Mount Hebo	1953	22	0.05	0.03	0.06	1st
Bald Mountain	1948	8	0.03	0.01	0.02	1st
Marys Peak	1948	11	0.02	0.29	0.04	2nd
Grass Mountain	1948	20	0.02	0.01	0.02	2nd
Prairie Peak—a	1948	24	0.06	0.06	0.08	1st
Prairie Peak—b	1948	20	0.04	0.02	0.05	1st

Image filtering, level slicing land cover classification, and matrixing

DOQs and rectified historical images were filtered using a 5×5 pixel focal minimum filter followed by a 5×5 low-pass filter. This filtering combination compensated for highly reflective (high brightness value) pixels within continuous forest canopy that would be misclassified during land cover classification as bald vegetation or bare ground unless their brightness values were reduced. The 5×5 focal minimum filter assigned the lowest brightness value of the neighboring 24 pixels to the central pixel. Focal filtering was followed by a 5×5 low-pass filter to deemphasize high-frequency edges (Lillesand et al. 2004).

Classification of pixels into three landcover classes (grass bald, forest, and bare ground) was accomplished by level slicing of gray-level brightness values. This method has previously been used on panchromatic images to classify pine forests and grasslands in the Colorado Front Range (Mast et al. 1997). Each filtered image was sliced individually, and brightness value ranges for each land cover class varied between each of the 10 images. Attempts to classify conifers and hardwoods separately were unsuccessful; instead a single forest land cover classification included both conifers and hardwoods. Using level slicing to classify the roads and buildings land cover class greatly reduced classification accuracy of the other three classes. Instead, the roads and buildings were manually delineated as polygons in separate shapefiles for each current and rectified historical image in ArcGIS 9.1 (2005 ERSI Inc.). Polygon shapefiles were converted to raster images and incorporated into each three-class classified image using an overlay function, resulting in four-class classified images.

Despite focal minimum and low-pass filtering, visual comparison of classified imagery to the geo-rectified images and DOQs found numerous instances

where small areas of forest were misclassified as grass balds. Clumping and sieving was used to reclassify these false balds as forest. An eight pixel window was used to clump contiguous groups of bald pixels into individual bald patches. Bald patches were determined manually by overlaying the clumped bald patches over the associated rectified image or DOQ. Clumped bald patches that were smaller than the smallest visually determined patches within a historical or DOQ image were reclassified as forest, resulting in historical and recent 4-class (forest, meadow, bare ground, and roads and buildings) land cover images for each study area. Change detection of bald land cover 4 between historical and recent classified images used a matrix function, creating a new image showing the coincidence of values between the historical and recent classified images. Each matrix classified image had 16 potential coincident classes (4 classes \times 4 classes). These classes were recoded, and the surface area of each land cover type was calculated from the number of pixels within each class.

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