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Quantifying regional trends in large live tree and snag availability in support of forest management

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ABSTRACT

In the Pacific Northwest of the United States, 20th century timber harvesting resulted in major declines in area of forests supporting large live and dead trees (i.e., snags), that are not only key habitat elements for many wildlife species but also critical components of ecosystem function. Regional forest management guidance, such as the Northwest Forest Plan (1994) and Eastside Screens (1995), may aim to conserve and foster the development of late-successional old-growth forests, characterized by large live and dead trees. Satellite remote sensing supports regional monitoring efforts of these habitat characteristics, but managers may require additional guidance in order to leverage these data for large landscape or regional assessments. In this study, our objectives were to assess long-term (historical vs. contemporary) and short-term (1993–2017) changes in lands supporting large live trees and snags across 10 wildlife habitat types (WHTs) – disparate vegetation conditions that support significantly different wildlife communities – in Oregon and Washington, USA. We generated 30-m, annual maps of large live trees (> 50 cm, > 75 cm, and > 100 cm diameter) and snags (> 25 cm and > 50 cm diameter) based on the gradient nearest neighbor (GNN) imputation method. GNN integrates Landsat satellite imagery, geospatial climatic and topographic data, and USDA Forest Service Forest Inventory and Analysis data to predict forest attributes for all forested lands in the study area. GNN classification accuracy was poor to good for large live trees (Cohen's kappa = 0.2 – 0.6) and poor to fair for snags (Cohen's kappa = 0.1 – 0.3) in most WHTs, though performance was substantially lower in drier WHTs where large live trees and snags were rare. Our results highlighted long-term reductions in forest supporting large live trees and snags from historical to contemporary times, especially in wetter, more productive WHTs. In contrast, we observed short-term (1993–2017) increases in areas supporting large live trees and snags. Federal forests were both more similar to reference conditions and exhibited greater recent increases in areas supporting large live trees compared to nonfederal lands. Thus, Oregon and Washington have lost a substantial proportion of forests containing large live trees and snags and recent recruitment of these trees at regional scales is a slow process primarily occurring on federal lands. However, detecting such changes through current Landsat satellite mapping technologies remains challenging, highlighting the need for new mapping methods to aid in future management.

1. Introduction

The presence of large live trees and standing dead trees, or snags, is a defining characteristic of old-growth forest ecosystems in western North America (Franklin et al. 1981, 2002, Kaufmann et al. 2007, Lindenmayer et al. 2012, Reilly and Spies 2015). They provide structural elements supporting high quality habitat for many wildlife species (Hunter and Bond 2001). Since the mid-20th century, anthropogenic stressors, such as timber harvesting, land conversion, and wildfire, have greatly reduced the extent of old-growth forests in Oregon and Washington (Bolsinger and Waddell 1993). Many wildlife species in this region rely entirely or in part on the availability of large live and dead trees (Ohmann et al. 1994) and monitoring their availability over short (i.e., decades) and long (i.e., centuries) time-scales may highlight trends in the capacity of landscapes to support them. Both the changes in late successional and old-growth abundance and their importance in supporting ecosystem function motivated regional management plans contributing to the conservation of these forests, such as the Northwest Forest Plan implemented in 1994 (Spies et al. 2019) and the Eastside Screens implemented in 1995 (Steen-Adams et al. 2017). Therefore,

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tools that integrate reference conditions with contemporary vegetation patterns and trends increasingly form the basis for regional and sub-regional monitoring programs (Davis et al. 2016) and support decision making associated with, for example, the management of forests for wildlife and biodiversity (Marcot et al. 2010).

Short- and long-term landscape changes are not expected to be spatially uniform, with variation within the region in the divergence from historical conditions and/or differing rates of change during recent decades. There can be substantial regional variation in the abundance of large live trees and snags, with environmental conditions contributing to these patterns. Local climatic and topographic factors constrain, for example, site productivity (Latta et al. 2010) and maximum tree height (Ryan and Yoder 1997, Koch et al. 2004, Swetnam et al. 2015). Tree mortality rates are constrained by environmental gradients, with higher mortality rates in high productivity forests (Franklin et al. 1987, Reilly and Spies 2016), resulting in greater abundance of snags. Broad-scale patterns in the frequency and severity of disturbance regimes impacts distributions of large live trees and snags by modifying successional trajectories (Reilly and Spies 2015) and contributing structural legacies to regrown stands.

Succession and management also affect snag densities, with a greater frequency in older, unmanaged stands or for several years following moderate or high severity wildfire. Recent (i.e., following the Northwest Forest Plan [1994] and Eastside Screens [1995]) management of federal forests in Oregon and Washington have emphasized promotion of late succession and old-forest attributes, including large snags (e.g.; Davis et al. 2015). Other management goals can conflict with the aim to increase snag densities, such as thinning to accelerate old forest conditions in moister forests (northwestern OR, western WA), and fuel reduction treatments in drier forests (southwestern OR, eastern OR and WA).

Integration of remote sensing, forest inventory, and spatially-distributed environmental data through multivariate analysis provides a method for assessing temporal trends in large live tree and snag abundance at relevant spatial scales (i.e., landscape/watershed scales). In Oregon and Washington, gradient nearest neighbor (GNN) imputation mapping – a method for integrating Landsat multispectral imagery with forest inventory data to produce wall-to-wall 30-m maps of forest attributes (Ohmann and Gregory 2002) – has become a standard tool for quantifying landscape pattern and change in forest ecosystems, especially as it relates to Northwest Forest Plan monitoring for both old-growth forest structure (Davis et al. 2015) and wildlife habitat (e.g., Davis et al. 2016). Uncertainty in pixel-level predictions of snag densities generated using GNN can be high and vary considerably across forest regions (Bell et al. 2015). Regional assessments of area within differing vegetation categories appear to be comparable to sample-based estimates for the same areas (Pierce et al. 2009, Ohmann et al. 2014). However, explicit assessment of bias and precision in estimates have been lacking. Consequently, map uncertainty impacts the degree to which managers can leverage remotely-sensed vegetation data for landscape to regional assessments of wildlife habitat.

Our main objective was to explore trends in the forestland area supporting large live trees and snags in the various wildlife habitat types distributed across Oregon and Washington, USA. Wildlife habitat types (WHTs; Table 1) represent macrohabitat groupings of vegetation communities by geographic distribution, physical setting, landscape setting, structure, and composition in Oregon and Washington shown to have similar patterns of use by breeding species of wildlife (O'Neil and Johnson 2001). Some wildlife habitat types are further divided into geographic subregions to take into account variation in dead wood amounts (Marcot et al. 2010) and wildlife responses to dead wood habitat features (J. Ohmann, personal communication; Mellen-McLean et al. 2017). As opposed to other classification schemes that rely solely on abiotic factors, such as physiographic region, WHTs were used to summarize our results because they account for current forest structure and composition and are thus more relevant to wildlife managers in the study region. We assessed long-term changes in large live tree and snag availability by comparing predicted historical reference conditions (i.e., properties of undisturbed ecosystems available for direct evaluation of natural ecosystem structure, composition, and function, similar to those found prior to Euro-American settlement) (Kaufmann et al. 1994, 2007) vs. contemporary forest area supporting large live trees and snags. We assessed short-term trends in large live tree and snag availability by examining interannual changes in mapped forest area supporting large live trees and snags following the initiation of the Northwest Forest Plan (1993) and Eastside Screens (1995).

Table 1
Description of wildlife habitat types (WHTs).

Wildlife Habitat Type	Abbreviation	Forest area (thousand ha)
Washington Coast[1]	WLCH_WCO	1729.8
Oregon Coast[1]	WLCH_OCO	2284.5
Washington Western Cascades[1]	WLCH_WCA	1894.5
Oregon Western Cascades[1]	WLCH_OCA	1258.2
Montane Mixed Conifer	MMC	3615.9
Southwest Oregon Mixed Conifer-Hardwood	SWOMC	1879.4
Lodgepole Pine Forests	LP	174.7
Northern Cascades and Rocky Mountains[2]	EMC_NCR	2305.8
Eastern Cascades and Blue Mountains[2]	EMC_ECB	2869.4
Ponderosa Pine/Douglas-fir	PPDF	1704.7

[1] Sub-regions of the Westside Lowland Conifer-Hardwood Forest.
[2] Sub-regions of the Eastside Mixed Conifer Forest.

2. Materials and methods

2.1. Study region

Our study area included all forestlands, regardless of ownership, in Oregon and Washington (Fig. 1). This region exhibits substantial variation in climate, topography, and ownerships (public and private), as well as various histories of disturbance, forest conditions, and departure from historical conditions (Spies et al. 2018). Forests range from wet temperate rainforests that rarely burned historically, but tended to be larger with mixed severity, to semi-arid woodlands that burned at high frequencies with generally low to moderate severity, resulting in substantial differences in species composition and forest structure. For this study, forestlands in the study region were divided into 10 WHTs (Table 1, Fig. 1), representing disparate vegetation conditions that support significantly different wildlife communities (O'Neil and Johnson 2001). Note that the prevalence of different land ownerships, and thus forest management policies, varies (Ohmann et al. 1994) with WHT as well, with greater private land ownership in the western portion of our study area (DeMeo et al. 2018).

2.2. Forest attribute mapping

Spatially complete and temporally extensive large live tree and snag density data are needed to assess their trends across forested landscapes. The GNN approach provides a regionally consistent, satellite-based approach to developing the necessary maps of forest attributes at annual resolutions. GNN is an imputation mapping methodology that relates geospatial data describing the environment to forest attributes measured at a subset of locations using a canonical correspondence analysis (CCA; Ter Braak 1986) to define distances between plots and pixels in a gradient space (Ohmann and Gregory 2002). These distances allow users to determine the most similar plots compared to a given pixel (i.e., k nearest neighbors) such that a prediction can be generated based on the measured forest attributes from the k nearest neighbors for each pixel (i.e., imputation). Here, we briefly describe GNN along with

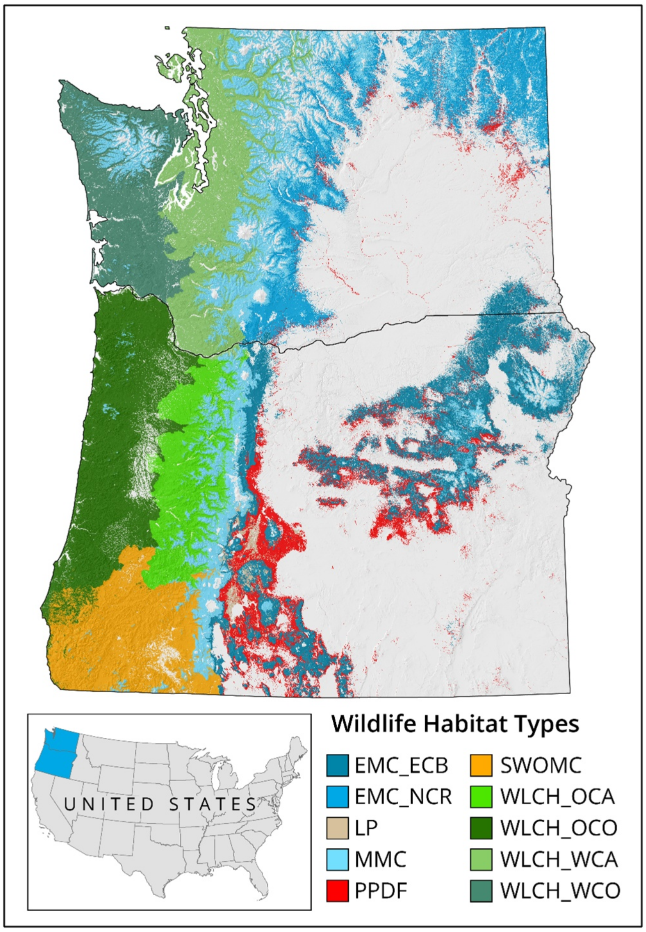

Fig. 1. Map of study area and wildlife habitat types. See Table 1 for wildlife habitat type names.

the relevant data for generating snag density maps.

2.2.1. *Forest attribute and geospatial data*

The mapping of forest structural and compositional data across broad landscapes requires both field-collected forest attribute data, Landsat imagery, and geospatial environmental predictors. Plot data for forest structural and compositional conditions, or forest attribute data, were collected on 34,488 forest inventory plots measured at 18,821 unique plots locations from 2001 to 2015. All plots shared a common sampling design and data collection protocol (Bechtold and Patterson 2005) and were measured by the U.S. Forest Service Forest Inventory and Analysis Program (FIA) and the Pacific Northwest Regional Biometrics Program. Inventory plots used a nested design with four subplots distributed across roughly one hectare. Individual live and dead trees measurements included diameter, height, and species. We used plots with at least 50% of their area classified as forest (i.e., at least 10% stocked) or forest-capable (evidence of previous forest and undeveloped for non-forest use, such as agriculture) (Bechtold and Patterson 2005). All plot-level forest attributes (e.g., live tree density and snag density by size-class) to be imputed were calculated on a per hectare basis for forested portions of the plot.

Spatially and temporally complete geospatial data were needed as inputs to GNN in order to produce mapped predictions for all pixels and years. We extracted both Landsat time series (LTS) imagery and environmental data (Table 2) for each forest inventory plot included in the species matrix to construct the environment matrix as an input into the CCA. Several factors, including cloud presence, sun angle, and phenology introduce substantial noise into LTS data. To reduce the impact of such noise on modeling, we used an ensemble LandTrendr disturbance mapping algorithm (Cohen et al. 2018) to produce temporally smoothed LTS imagery (Ohmann et al. 2012, Kennedy et al. 2018). LandTrendr is a trajectory-based change detection method that identifies fitted line segments of consistent trajectory for each pixel that describe sequences of disturbance and growth. Traditionally, segments are delineated using a single spectral band or index, but vertices are applied across all bands and indices. Cohen et al. (2018) modified this approach to generate multiple versions of LandTrendr, each using a different band/index to delineate segment endpoints. These multiple Landtrendr versions are used as input in a random forest model (Breiman 2001) along with reference pixel trajectories derived using TimeSync, a visual image interpretation tool (Cohen et al. 2010) – see Cohen et al. (2018) for additional details. As with traditional LandTrendr, the output of the ensemble LandTrendr is a set of vertices for each pixel that can be used to create the temporally smoothed LTS imagery. From the temporally smoothed LTS imagery, we extracted tasseled cap indices (Crist and Cicone 1984) used in Ohmann et al. (2014) and the normalized burn ratio (Key and Benson 2006).

Table 2
Geospatial data used in LandTrendr-GNN Framework.

Subset	Code	Description
Landsat	TC1	Brightness (i.e., axis 1 of the tassel cap transformation – unitless)
	TC2	Greenness (i.e., axis 2 of the tassel cap transformation - unitless)
	TC3	Wetness (i.e., axis 3 of the tassel cap transformation - unitless)
	NBR	Normalized burn ratio (unitless)
Climate	ANNPRE	Mean annual precipitation (ln[mm])
	ANNTMP	Mean annual temperature (°C)
	AUGMAXT	Mean maximum temperature of August (°C)
	DECMINT	Mean minimum temperature of December (°C)
	SMRTP	Ratio of mean temperature (°C) to precipitation (ln [mm]) of May-Sept.
Location	COASTPROX	Optimal path length from the coastline (km)
	LAT	Latitude (°)
	LON	Longitude (°)
Topography	ASPTR	Cosine transformation of aspect
	DEM	Elevation from a digital elevation map (m)
	PRR	Potential relative radiation (unitless)
	SLPPCT	Slope (%)
	TPI450	Topographic position index, difference between elevation and mean elevation within a 450-m radius window

Past experience indicates that, depending on the physiographic region being modeled and mapped (hereafter, modeling region), the relationship between LTS data and forest structure depends on other environmental variables, such as climate, topography, and geographic location (Ohmann et al. 2014). Climate data were derived from the Parameter elevation Relationships on Independent Slopes Model (PRISM) using 30-year normals (1981–2010) of mean monthly precipitation and temperature data (Daly et al. 2008), an 800-m data product that we resampled to 30-m pixels using bilinear interpolation. Topographic variables were derived from 10-m digital elevation models and resampled to 30-m before computing derivative products (e.g. slope percent, aspect).

2.2.2. *CCA modeling and gradient nearest neighbor imputation*

To construct large live tree and snag presence maps, we imputed live tree and snag density (ha^{-1}) by size class to all 30-m pixels using the GNN methodology. We provide a brief description of GNN here, but detailed descriptions of GNN are available in our previous publications (Ohmann and Gregory 2002, Ohmann et al. 2014, Davis et al. 2015, Kennedy et al. 2018). A matrix of forest attributes based on the reference data (i.e., the species matrix) was constructed using species and size-class specific live tree basal area. Previous regional forest mapping with GNN has used a species matrix containing live tree basal area separated by species and tree size class to represent both compositional and structural variation in forest ecosystems (Ohmann et al. 2012, Davis et al. 2015). A matrix of geospatial data (i.e., the environment matrix) was constructed from Landsat, climate, topographic, and location information (Table 2). To create the environment matrix, we extracted all geospatial data for the 90-m by 90-m (i.e., 3-by-3 pixel) footprint for each plot during the year of measurement to avoid a temporal mismatch between plot measurement and LTS imagery acquisition (other geospatial variables were assumed constant during the study period.). FIA subplots and macroplots cover 8% and 50% of the 3-by-3 pixel area, respectively. Our use of the full plot and the 3-by-3 pixel area minimizes the impacts of geolocation errors (Zald et al. 2014) and spatial heterogeneity in forest attributes which may make the analysis of individual subplots untenable (Stehman and Wickham, 2011).

A subset of forest inventory plots was removed from the sample to minimize the impacts of outliers on modeling and mapping. Most often, plots were removed from the dataset during screening because they straddled multiple forest conditions for which neither the forest attribute data nor the LTS data were comparable (e.g., old-growth and plantation forest). Plots were also removed if a disturbance occurred between the date of forest inventory measurement and the date of LTS acquisition. All remaining plots (21,721 total plot measurements at 14,566 unique locations), regardless of measurement year, were partitioned by modeling region and the associated modeling region species and environmental matrices were used as the basis for CCA modeling. Neighbor-finding was based on weighted Euclidean distance within multivariate gradient (CCA) space, with axis scores weighted by their eigenvalues. This approach assumes that the relationship between forest attributes and geospatial data is stationary through time.

2.2.3. Pixel classification

To characterize the frequency of forestlands that support large live trees and snags, we generated classified maps for the presence of live trees ≥ 50 cm, live trees ≥ 75 cm, live trees ≥ 100 cm, snags ≥ 25 cm, and snags ≥ 50 cm. These size categories were selected to represent the variation and complexity in important habitat based on life history requirements for a wide array of wildlife species (Edworthy et al. 2018, Sandström et al. 2019). For example, cavity nesting birds may require large snags (Hayes and Hagar 2002), and thus be limited to those habitats where those large snags are present. Such patterns can be further complicated is habitat requirements change throughout an animals lifecycle.

Each pixel was classified as present or absent for a given group based on whether the majority of the seven nearest neighbors ($k = 7$) indicated that trees or snags of a given size were present. By using seven nearest neighbors, we minimize spatial noise in mapping while retaining the importance of relatively rare forest conditions on the landscape (Battles et al. 2018, Davis et al. *in review*). For comparisons with individual plots, presence or absence was determined by taking the majority classification for the nine pixels coincident with the plots locations (i.e., 3-by-3 pixel footprint). We assessed map classification accuracy using error matrices, summarizing the results for each WHT as well as all forestland for each of the five large live tree and snag classes. Specifically, we generated overall accuracy (percentage of plots correctly classified) and Cohen's kappa (a measure of prediction accuracy as compared to random assignment), with greater values indicating better performance in each case (Fielding and Bell 1997). In the case of Cohen's kappa, values may represent poor (< 0.4), good (0.4–0.75), and excellent (> 0.75) classification performance at the plot-scale (0.81 ha).

2.2.4. Reference conditions

In addition to vegetation mapping, we estimated the forest area supporting large live trees and snags under reference conditions. Reference conditions are properties of undisturbed ecosystems available for direct evaluation of natural ecosystem structure, composition, and function, and are often assumed to be similar to those found prior to Euro-American settlement (Kaufmann et al. 1994, 2007). Thus, reference conditions reported in this study were assumed to appropriately characterize conditions prior to major changes in disturbance regimes during the 19th and 20th centuries (e.g., timber harvesting, fire suppression, etc.) and can therefore be compared with contemporary conditions to assess changes in large live tree and snag prevalence following Euro-American settlement.

For each plot, we determined which plots had not experienced human-caused disturbances (hereafter, reference plots). We used information recorded by inventory field crews as part of field-sampling protocol to indicate presence or absence of signs of tree-cutting. Plots without any sign of tree-cutting were classified as unharvested. For each reference plot, we used the tree data to classify them as early- (dominated by trees < 25.4 cm diameter or open with remnant larger trees), mid- (dominated by trees 25.4 cm – 50.8 cm diameter), or late-seral forest (dominated by trees > 50.8 cm diameter) (Ohmann et al. 2017). Rules for assigning seral status varied by WHT, and employed varying combinations of stem density and canopy cover of trees of different diameter ranges (Ohmann et al. 2017). This reflects the fact that the relationship between forest structure and forest age or seral state is expected to vary as a function of tree species composition and biophysical setting. We used estimates of the proportion of forestland in early-, mid-, and late-seral conditions based on state and transition models to develop area-weighted factors for determining landscape-scale large tree and snag density distributions (i.e., proportion of landscape in a given seral state for each WHT). The state and transition models were developed under the LANDFIRE program (https://www.landfire.gov/), and were assembled by Haugo et al. (2015) for a regional assessment of forest restoration needs. We linked these models to the map of potential vegetation from the Integrated Landscape Assessment Project (Halofsky et al. 2014) to allow spatial weighting within WHTs.

Finally, we constructed empirical cumulative density functions for the density of large live trees and snags for each WHT under reference conditions (Fig. 2). We ranked reference plots by large live tree or snag density and multiplied the frequency of each tree density value by the proportion of the total plot in the seral status, the area that each plot represents based on the sampling design, and the weights derived from the state and transition models for the associated seral state. We weighted the frequency of tree densities in our reference plot pool by the modeled historical frequency of seral states to account for biases in the contemporary reference plots, which tend to be located at higher elevations and on public lands. Thus, the relative frequency of forestlands with certain densities of large live trees or snags under reference condition similar to those predicted by the state and transition models was represented by the empirical density function. For our analysis, we focused on lands supporting any number of large live trees or snags (i.e., density greater than 0), but the empirical density function could also be used to identify specific habitats associated with the historical landscape, such as stands with high densities of large live trees and snags that might provide rare, but important habitat.

2.3. Assessing large live tree and snag trends

In this study, we examined both long-term and short-term trends in the area of forestland supporting large live trees and snags by WHT and ownership (federal vs. nonfederal). To examine long-term trends we compared error-adjusted estimates of area (Olofsson et al. 2013, 2014, Stehman 2013) derived from classified GNN maps with estimates from our reference conditions. The error-adjusted estimator of area both corrects for biases in the mapping and provides a method for calculating confidence intervals in those areas. It relies on the error matrix from the comparison of map-based and plot-based classifications.

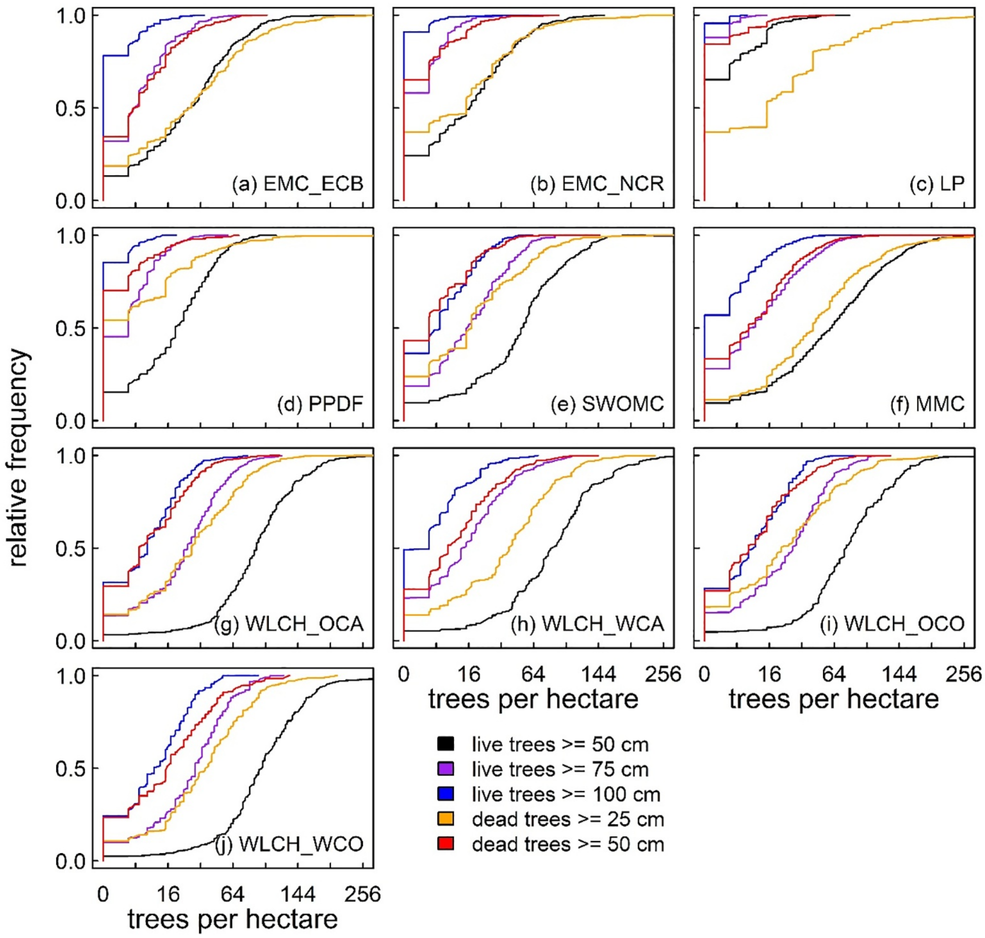

Fig. 2. Empirical weighted cumulative density functions for reference condition densities of trees and snags above diameter thresholds for the 10 wildlife habitat types.

Therefore, we used GNN maps and the most recent measurements of the FIA plots (2007–2016) to generate error-adjusted area estimates representing the mid-point of the measurement cycle (i.e., 2011). A subset of plots used to develop the GNN mapping were collected as an intensification of the inventory on US Forest Service lands. These plots from the US Forest Service intensification were excluded because they focus on National Forest System lands and their inclusion here could bias the error adjustment procedure. Finally, we rescaled the 2011 error-adjusted area estimates and the reference condition estimates (Section 2.2.4) as a function of total forestland (i.e., percent of forestland supporting large live trees or snags) to facilitate comparisons both within and across WHTs.

For short-term trends, we calculated area by year and wildlife type for each large live tree and snag density category. In order to simplify the assessment of gains or losses in area, we calculated the annual rates of change in the area supporting large live trees or snags since 1993. We selected 1993 as the baseline year to immediately precede the initiation of the Northwest Forest Plan, which is also similar to the initiation of Eastside Screens (1995). Trends in area reported in this study were ingested into a web-based trend analysis tool and can be viewed and downloaded at https://lemma.forestry.oregonstate.edu/trend/wht. The web-based trend tool represents an technological advance in dissemination of cutting edge mapping results that minimizes the GIS expertise required to interact to forest monitoring data.

3. Results

Empirical density functions for the relative frequency of lands with differing densities of large live trees and snags varied by WHT (Fig. 2). As the diameter threshold for large live trees and snags increased, the frequency of reference plots with no such trees increased and the maximum densities decreased, reflecting increasing scarcity of large live trees and snags. Both the relative frequency with no large trees increased and maximum densities decreased as one shifted from wetter western WHTs (SWOMC, MMC, WLCH_OCA, WLCH_WCA, WLCH_OCO, and WLCH_WCO) to drier, eastern WHTs (EMC_ECB, EMC_NCR, LP, and PPDF). In particular, large live trees and snags were rare for LP and, to a lesser extent, PPDF.

Overall accuracy and Kappa varied by WHT, tree status (live vs. dead), and tree size (Fig. 3). When compared to eastern WHTs (EMC_ECB, EMC_NCR, LP, and PPDF; Table 1), western and high elevation WHTs (SWOMC, MMC, WLCH_OCA, WLCH_WCA, WLCH_OCO, and WLCH_WCO) exhibited greater Kappa statistics for all large live tree classes (0.4–0.6 vs. 0.0–0.3), greater overall accuracy for live trees greater than 50 cm and 75 cm thresholds (75%–82% vs. 61%–75%), and lesser overall accuracy for live trees greater than 100 cm (76%–86% vs. 85%–98%). Interestingly, two WHTs (LP and PPDF) exhibited high overall accuracy for the largest live tree and snag categories (83%–98%), but low Kappa statistics associated with poor performance (−0.03–0.09). Both overall accuracy and Kappa tended to be greater for live tree vs. dead tree classification and generally increased when considering larger vs. smaller trees and snags. Overall accuracy for classification based on the presence of large live trees ranged from 61% to 81%, 65% to 82%, and 76% to 98% for 50 cm, 75 cm, and 100 cm diameter thresholds, respectively. Kappa based on the presence of large live trees ranged from 0.15 to 0.60, 0.17 to 0.50, and 0.00 to

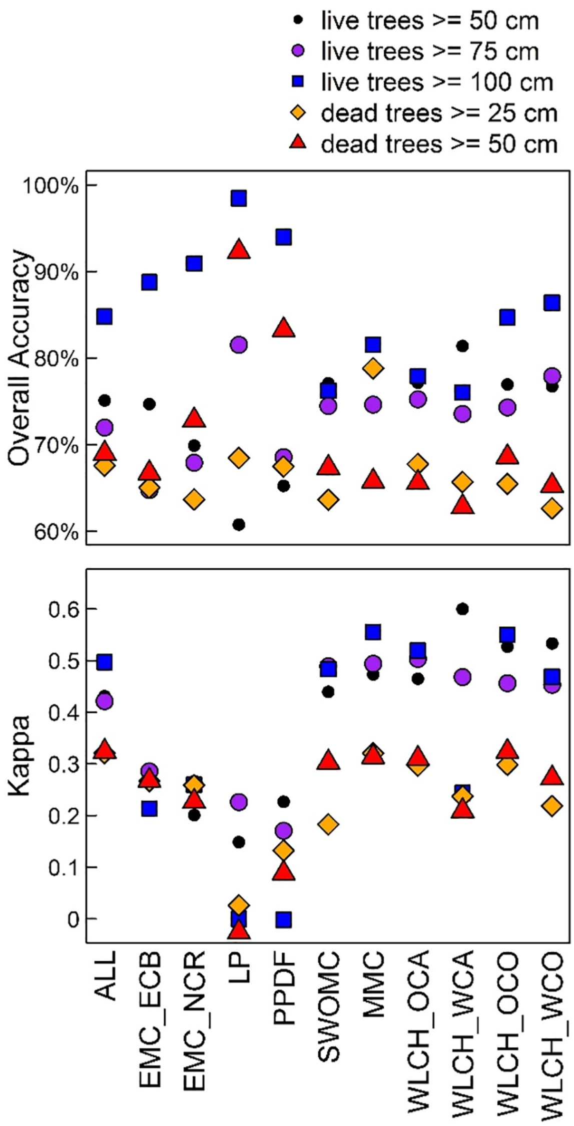

Fig. 3. Overall Accuracy and Kappa statistics for large live tree and snag classification.

0.55 for 50 cm, 75 cm, and 100 cm diameter thresholds, respectively. Overall accuracy for classification based on the presence of snags ranged from 63% to 79%, and 63% to 92% for 25 cm and 50 cm diameter thresholds, respectively. Kappa based on the presence of snags ranged from 0.03 to 0.32, and −0.03–0.32 for 25 cm and 50 cm diameter thresholds, respectively.

The error-adjusted area estimates for 2011 indicated that, in most cases, current forest area supporting large live trees and snags was less than historical reference conditions (Fig. 4, Table 3). The differences between historical and current forest area supporting large live trees or snags were generally greater for forests in the western portion of the study area (WLCH_OCA, WLCH_WCA, WLCH_OCO, WLCH_OCO, MMC, and SWOMC) compared to eastern WHTs (EMB_ECB, EMC_NCR, LP, and PPDF). Reference forest area supporting large live trees was within the contemporary 95% confidence intervals for current conditions in LP (all size classes) and EMC_NCR (≥100 cm). Federal lands had a greater percentage of forest area supporting large live trees and snags compared to nonfederal lands. In some cases, current conditions did not differ (within confidence intervals) from reference conditions. Almost all of the examples of current conditions not differing from historical reference conditions were observed for federal lands, with the examples on nonfederal lands being associated with the LP WHT.

While individual WHTs indicated substantial regional variation in the contemporary temporal trends in forestlands supporting large live trees and snags (1993–2016; Fig. 5), the mean annual rates of change indicated a general increase over time since 1993 (Fig. 6). The mean across WHTs in the annual change in area supporting large live trees and snags increased by an average of −0.002%-0.091% and 0.006%-0.029%, respectively, depending on the size category and ownership. For all lands, 95% confidence intervals were greater than zero for live trees ≥ 50 cm and snags (≥25 cm and ≥ 50 cm) and 68% confidence intervals were greater than zero for all classes except forestlands with live trees ≥ 100 cm. Federal lands showed increases (95% confidence intervals greater than zero) for all classes except forest with live trees ≥ 100 cm (only 68% confidence intervals greater than zero). In contrast, chnage on non-federal lands were not different from zero (95% and 68% confidence intervals included zero) in most cases, with marginal exceptions for forest supporting snags ≥ 50 cm (68% confidence intervals greater than zero). Thus, federal forestlands during recent decades account for much of the regional trends in forest area supporting large live trees and snags.

4. Discussion

Contemporary patterns of large live tree and snag distributions implied an important role for regional patterns of 19th and 20th century disturbance legacies, including widespread timber harvesting and historic wildfires, as well as contemporary land management. Forests in the wetter and warmer portion of the study region (WLCH_OCA, WLCH_WCA, WLCH_OCO, WLCH_OCO, and SWOMC; Table 1, Fig. 1), generally contained 20% to 60% of the forested area supporting large live trees and snags compared to historical reference conditions, with differences being even more extreme on nonfederal forestlands (Fig. 4). These results reflect the extensive conversion of late-successional and old-growth forest to second-growth plantations, especially on private lands (Bolsinger and Waddell 1993, Spies et al. 2018). In contrast, contemporary and historical forest area supporting large live trees and snags tended to differ by lesser magnitudes in drier portions of the study area and on federal lands (EMB_ECB, EMC_NCR, LP, PPDF; Table 1, Fig. 1). For example, reference and contemporary forest area supporting large lodgepole pine trees were not significantly different. Smaller relative differences between contemporary and reference condition patterns may reflect regional differences in timber harvesting practices (clearcutting vs. selective removal of large trees) (Spies et al. 2018) or the scarcity of some large tree elements in historical reference conditions (Figs. 2 and 4). Due to the recognition of the importance of snags for wildlife habitat and their paucity in the Pacific Northwest due to decades of intensive forest management, snag creation is a contemporary practice needing critical evaluation (Barry et al. 2017).

While we observed increases in areas supporting large live trees and snags during recent decades (Fig. 6), these changes in area were relatively small (Fig. 5) and are therefore unlikely to offset past losses for the region as a whole. We observed increases in area with large live trees and snags on federal lands, but not on nonfederal lands, indicating that those federal lands drive most of the regional changes. The reduction of forest area supporting high densities of large live trees and snags since historical times and the lack of substantial increases during recent decades on all lands highlights major challenges to maintaining quality habitat for some wildlife species. Large, continuous tracts of suitable old-growth forest habitat are already restricted by historical land-use history and further endangered by emerging issues, like climate change and wildfire (Davis et al. 2015, 2016, Phalan et al. 2019). For example, wildfires may reduce live tree densities and increase snag densities through tree mortality and may reduce snag density through consumption of pre-existing snags. Therefore, we expect continued changes in the prevalence of forest lands supporting large live trees and snags which are important for managing wildlife habitat and biodiversity. Coupled with our observation that most forests supporting large

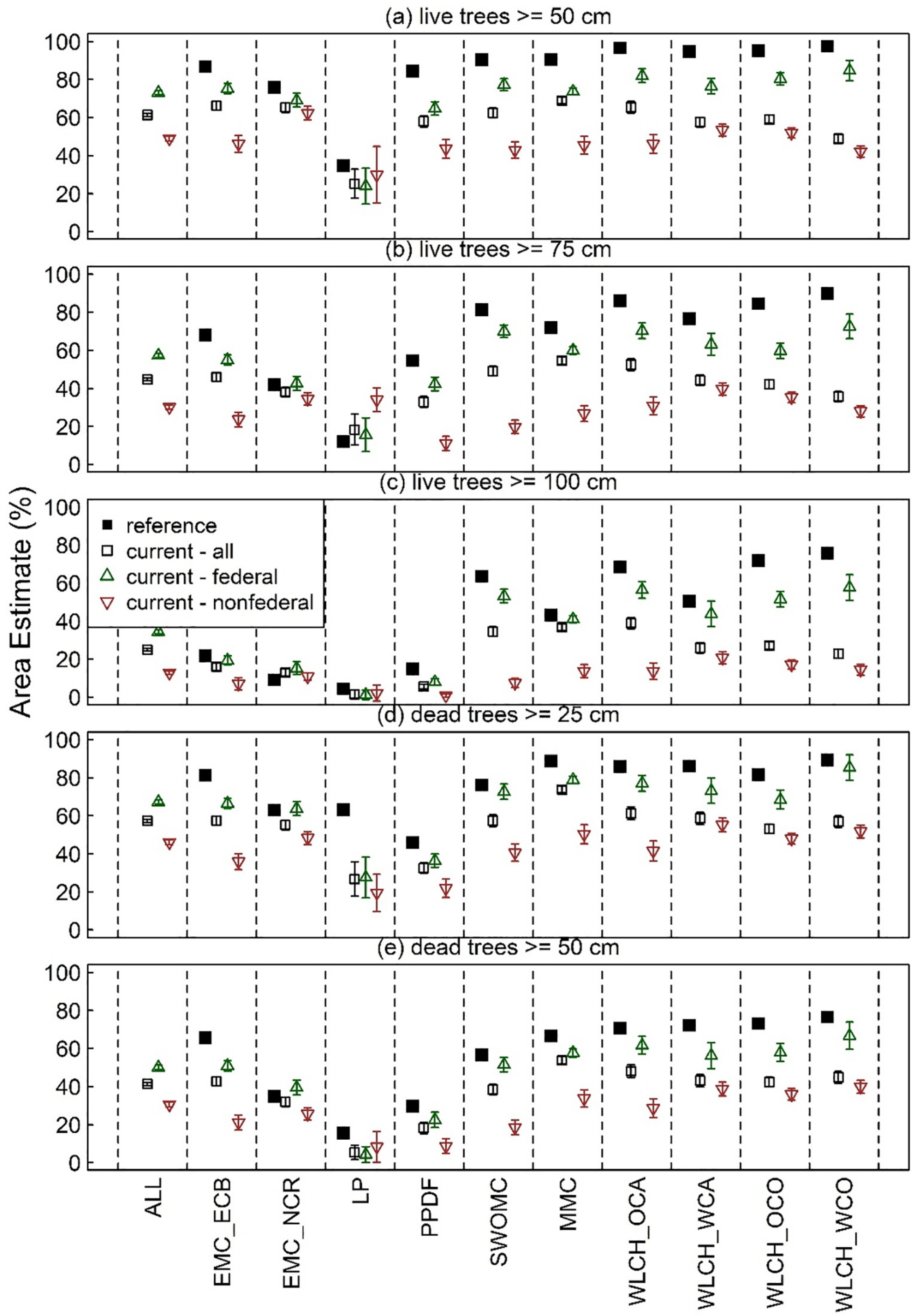

Fig. 4. Comparison of current (2011 mid-point) (with 95% confidence interval) and historical estimates for the percentage of forest area supporting large live trees or snags (Fig. 2).

live trees and snags are federally managed (Fig. 4), area change results indicate that federal forest lands have and will continue to play a dominant role in the management and conservation of large live trees and snags in the region.

Still, it is important to note challenges presented by monitoring change in forest ecosystems based on satellite remote sensing. The accumulation of large live trees and snags may simply be a slow process poorly captured by the current study. The disparity between the rates of development and attrition of large live trees and snags suitable as habitat elements for wildlife species (e.g., Pacific marten) implies that these structures should be considered in forest management (Delheimer et al. 2019). Slow and steady changes in forest structure occurring in stable (i.e., undisturbed) forests may be difficult to represent with our satellite based approach, which is best suited for disturbance-mediated changes (Battles et al. 2018, Kennedy et al. 2018). As a result, our methods may under-predict accumulation of large live trees and snags in undisturbed forests. In part, the challenge of assessing change through differencing two classified maps may be similar to the challenges in estimating change by differencing mean population states from forest inventory plots for two points in time in that errors in flux estimates can be greater than the magnitude of the flux itself (e.g., Fried and Xiaoping 2008). Large live tree and snag mapping would certainly

Table 3
Forest area (1000 s ha) and percent forest area classified supporting large live trees and snags within each wildlife habitat type in 2016.

Wildlife Habitat Type	Forest area with trees exceeding diameter threshold (1000 s ha)									
	Live trees						Dead trees			
	≥ 50 cm		≥ 75 cm		≥ 100 cm		≥ 25 cm		≥ 50 cm	
	1000 s ha	%	1000 s ha	%	1000 s ha	%	1000 s ha	%	1000 s ha	%
EMC_ECB	2070	72	1290	45	337	12	1655	58	1078	38
EMC_NCR	1586	69	808	35	208	9	1297	56	651	28
LP	81	47	50	29	11	6	51	29	26	15
PPDF	1066	63	535	31	99	6	576	34	326	19
SWOMC	1234	66	934	50	637	34	1128	60	720	38
MMC	2530	70	1864	52	1159	32	2770	77	1915	53
WLCH_OCA	810	64	636	51	444	35	792	63	601	48
WLCH_WCA	1061	56	750	40	403	21	1143	60	793	42
WLCH_OCO	1309	57	910	40	600	26	1297	57	961	42
WLCH_WCO	752	43	507	29	309	18	1033	60	782	45
All	**12,501**	**63**	**8283**	**42**	**4205**	**21**	**11,742**	**60**	**7852**	**40**

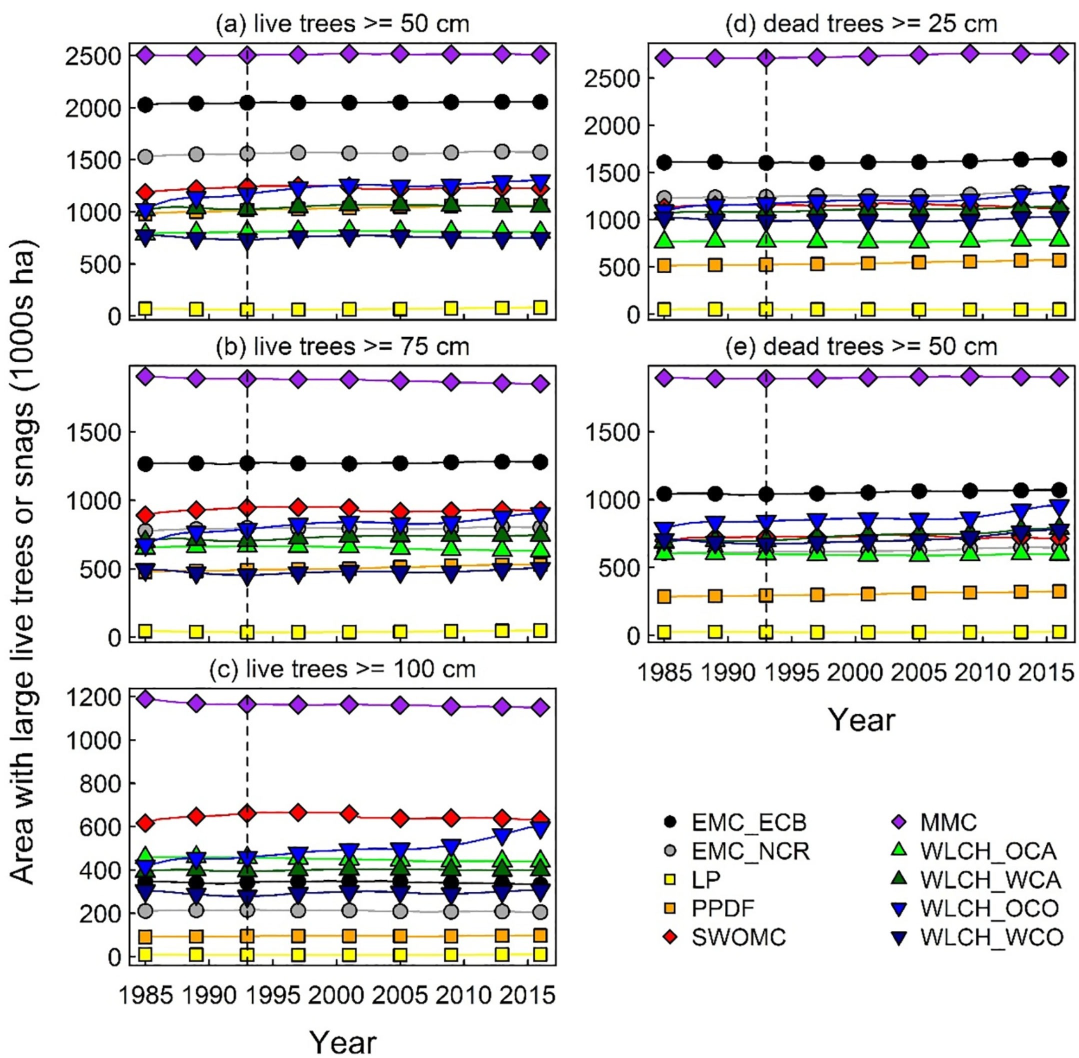

Fig. 5. Trends in forest area supporting large live trees or snags for each of 10 wildlife habitat types (WHTs). The vertical dashed line indicates 1993 and coincides with the initiation of two regional forest management frameworks: the Northwest Forest Plan and the Eastside Screens.

be improved by forest height information (Zald et al. 2014), indicating the regional aerial or satellite lidar acquisitions might lead to improved maps.

Classification performance for lands supporting snags was poor (kappa −0.03–0.32) for large snags and poor to good (kappa ranging from 0.2 to 0.6) for large live trees (Fig. 3), implying substantial uncertainty in plot-level predictions of large live tree and snag occurrence patterns. Accuracy declined in the dry, eastern WHTs with less forest

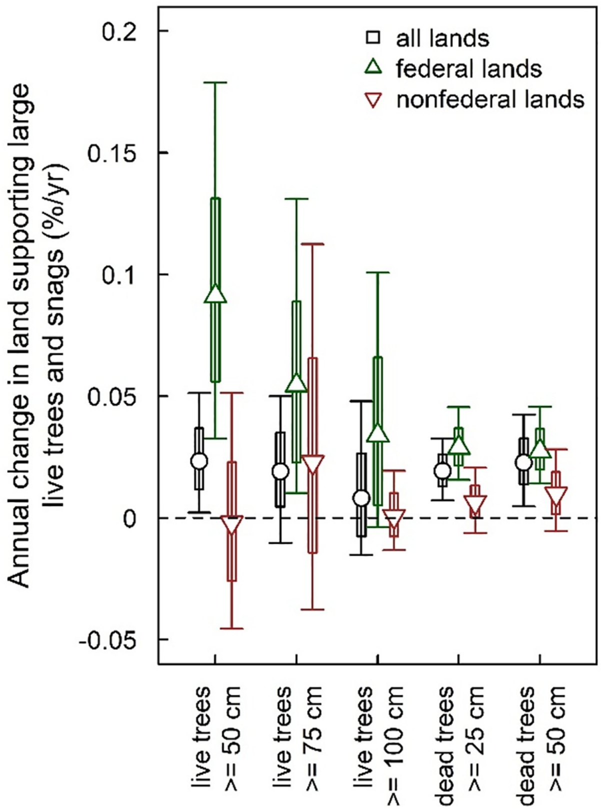

Fig. 6. Mean annual rates of change (1993–2016) in land supporting large live trees and snags on all lands, federal lands, and nonfederal lands, with 68% and 95% confidence intervals. Symbols indicate the mean change, boxes indicate the 68% confidence intervals, and whiskers indicate the 95% confidence intervals.

supporting large live trees and snags, implying that model performance is likely to decline when trying to predict the occurrence of rare forest components. While plot-level GNN prediction accuracy may be low for some attributes, as shown for snag densities in this paper as well as our previous work (Bell et al. 2015), assessments based on GNN maps at landscape (Ohmann et al. 2014) and regional (Pierce et al. 2009) scales can still perform well. Thus, regardless of accuracy, it is most appropriate to utilize these maps for strategic planning and monitoring at landscape to regional scales. Despite these uncertainties, providing forest trend data through flexible tools, such as web interfaces (Fig. 5), supports monitoring and planning activities essential for forest management and conservation, albeit at landscape- to regional-scales.

In contrast, the comparisons to historical reference conditions were likely insensitive to classification accuracy. For estimating area for a single point in time, the error-adjusted area estimation procedure addressed classification error explicitly, correcting estimates based on the design-based sampling itself. However, even after accounting for classification biases, contemporary vs. historical reference condition comparison (Fig. 4) remain challenging. To define historical reference conditions, we used contemporary measurements of undisturbed forests and predictions of the historical frequency of differing seral states to derive reference conditions for the density of large live trees and snags. However, forests in some WHTs may have dramatically changed since pre-settlement. For example, fire exclusion has increased tree densities in ponderosa pine forests, potentially increasing the density of trees in smaller size classes above historical levels (Spies et al. 2018). Thus, the reference conditions themselves also incorporate error. Given that comparisons for small size class thresholds (i.e., ≥25 cm for snags and ≥ 50 cm for live trees) were qualitatively similar to results for larger tree size classes (Figs. 4-6), it appears that errors in reference conditions did not substantially alter our conclusions.

5. Conclusions

In part, the maintenance of forest biodiversity depends on the retention of existing and restoration of late-successional and older forest habitat for wildlife species to within natural or desired ranges. Our results indicate that the forests of Oregon and Washington have lost a substantial proportion of forests supporting large live trees and snags compared to reference conditions and that recruitment of these structural elements of wildlife habitat at regional scales is a slow process. Slow accumulation of old forests will alter habitat availability and impact wildlife across the region in varying ways depending on the species habitat requirements and other stressors and threats. Large quantities and diversities of dead wood structures are needed to maintain biodiversity in forest ecosystems (Sandström et al. 2019). Given that most forests supporting large live trees and snags are federally managed, these federal lands will be central to the conservation and management of large live trees and snags in the Pacific Northwest. Furthermore, our results indicate that our capacity to detect such changes through current regional mapping technologies (e.g., satellite-based imputation modeling) may be somewhat limited, especially for snags that occur in older stable forests. Even when plot-level predictive accuracy is low, as observed for snag mapping here and elsewhere (Bell et al. 2015), aggregation of pixel-level predictions up to landscape- or regional-scale estimation appears robust (Pierce et al. 2009, Ohmann et al. 2014). Therefore, application of satellite-based remote sensing for habitat assessment appears most appropriate at landscape- to regional-scales, even when plot-level accuracies are poor. Still, making maps of forest vegetation freely and easily available, as we have done through our web-based trend tool, puts information in the hands of those who need it. Therefore, research is needed to develop more reliable methods for modeling, mapping, and monitoring changes in key ecosystem characteristics of stable older forest ecosystems that change slowly through time, such as aboveground live biomass (Battles et al. 2018) or large live trees and snags densities.

CRediT authorship contribution statement

David M. Bell: Conceptualization, Methodology, Formal analysis, Writing - original draft, Writing - review & editing, Visualization, Supervision, Project administration. **Steven A. Acker:** Conceptualization, Methodology, Writing - review & editing. **Matthew J. Gregory:** Conceptualization, Methodology, Formal analysis, Writing - review & editing, Visualization. **Raymond J. Davis:** Conceptualization, Writing - review & editing. **Barbara A. Garcia:** Conceptualization, Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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