A New Pest in the West: Simulating Emerald Ash Borer-Related Ash Mortality and Plant Community Shifts in the Pacific Northwest

by
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ABSTRACT

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Oregon ash (*Fraxinus latifolia*), the sole native species of ash tree in the Pacific Northwest, is under threat due to the recent arrival of the emerald ash borer (*Agrilus planipennis*; EAB) near Portland, OR in 2022. EAB has already decimated ash populations across the Midwest and East Coast following its introduction to the United States in Michigan in 2002. It is now expected to have similar impacts in the Pacific Northwest, posing damage to the sensitive riparian and wetland habitats that Oregon ash occupies. Given the new invasion, this research aims to address two questions: 1) how fast will EAB spread across Oregon and Washington over the next 25 years and how might this affect the spatial pattern of Oregon ash mortality? 2) how might plant community successional trajectories be altered in areas affected by EAB? This study used LANDIS-II, a spatially explicit forest landscape model, to project the spatial pattern of EAB-caused ash mortality in the Pacific Northwest and its effects on successional trajectories. Due to the large variation in EAB dispersal rates in the literature, three EAB dispersal rates with and without human-aided EAB dispersal via wood transportation were tested. Results indicate that a conservative, low EAB dispersal rate (0.4 km/year) restricts EAB to northwest Oregon through 2050, while a high EAB dispersal rate (10 km/year) allows EAB to reach the southern extent of the landscape by ~2040 and the northern by ~2050. As expected, adding long-range human dispersal events accelerates the spread of EAB across the landscape and ensuing Oregon ash mortality. After EAB disturbance, western redcedar was predicted to increase in biomass across the landscape while other species co-occurring with Oregon ash exhibit mixed increases and decreases. Despite these increases in biomass, pre-disturbance total biomass was not predicted to recover by 2050. Understanding these ranges of possible Oregon ash futures is important for strategic planning efforts of local land managers and owners as they prepare for high levels of ash mortality and develop plans for restoration.

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INTRODUCTION

The combined forces of globalization and climate change are causing both exotic and native forest pests to inhabit new areas and infest new hosts (Ramsfield et al., 2016). Global trade networks have allowed forest pests to travel long distances and attack susceptible tree species, however, these invasive insects are not equally destructive nor prevalent. While detection of some invasive insect types has remained steady since 1850 in the United States, detection of wood- and phloem-boring insects has increased since 1980, possibly reflecting improved surveying and increases in wood packing material use (Aukema et al., 2010). Furthermore, climate change is simultaneously shifting host tree distribution and pest distribution, allowing some pests to increase their range in latitude and elevation (Ramsfield et al., 2016). This often results in increased habitat area for forest pests and therefore potentially more overlap with host tree ranges (Ramsfield et al., 2016).

Tree pests can have a wide variety of negative impacts on forest communities and their ecological services through reductions in biodiversity, carbon storage, flood mitigation, and recreational and cultural value (Boyd et al., 2013). The impacts extend beyond the ecological; Aukema et al. (2011) estimated that non-native wood- and phloem-boring insects are responsible for $1.7 billion of municipal government spending and $830 million of lost residential property value annually.

The emerald ash borer (*Agrilus planipennis*; EAB) is an exotic beetle that attacks all species of ash trees (*Fraxinus spp.*) (Baranchikov et al., 2014). EAB has also been found to infest two less preferred hosts, cultivated olive trees (*Olea europaea*) and white fringetrees (*Chionanthus virginicus*) (Cipollini, 2015). This beetle is native to temperate northeastern Asia, where healthy ash trees are resistant to EAB damage and only stressed or dying ash trees are susceptible (Baranchikov et al., 2014). EAB kills ash trees when its wood-boring larvae feed on the tree's inner bark, essentially girdling the tree (Haack et al., 2015). Larvae spend one to two years feeding on the cambium, creating S-shaped galleries, overwinter as pupae, and emerge in spring or early summer through distinctive D-shaped exit holes (Haack et al., 2015). EAB flight capabilities outside of its native range are not well understood, but the mature beetles have been estimated to fly as little as 0.4 km (Mercader et al. 2016) and up to 10 km (Taylor et al., 2010) to a new host tree where they spend a month feeding on ash leaves and mating. Additionally, long distance EAB dispersal (LDD) can occur when people unknowingly move infested ash wood

products (e.g., firewood) (Cappaert et al., 2005). Mature EAB females then lay their eggs in the crevices of ash tree bark where they hatch and chew through the bark to the cambium, beginning the cycle over again (Poland et al., 2015).

The first documented detection of emerald ash borer in the United States occurred in Michigan in 2002, although dendrochronological records indicate that it arrived in the early 1990s (Haack et al., 2002; Siegert et al., 2014). Since EAB's arrival to the United States, which likely occurred via wood packing material shipped from its native range, EAB has decimated ash populations in the midwestern and eastern United States and is now present in 37 states and six Canadian provinces as of 2026 (Emerald Ash Borer Information Network, 2026). Although there are no published nationwide estimates of EAB-related ash mortality since its introduction, ash mortality exceeded 99% near invasion epicenters in Michigan's Huron River Watershed (Klooster et al., 2014).

EAB was observed for the first time on the West Coast in 2022 near Portland, Oregon and since then has spread across five counties in northwest Oregon (Oregon Department of Agriculture [ODA] & Oregon Department of Forestry [ODF], n.d.). Oregon ash *(Fraxinus latifolia*), the Pacific Northwest's sole native species of ash tree, is now facing inevitable decline and potential extinction due to the recent arrival of EAB within its range. Since 2022, EAB has been found on 150 ash trees in the greater Portland area. These infested trees are primarily around two main epicenters in Forest Grove and Woodburn, OR, however, three new epicenters near eastern Portland, Aloha, and Oregon City, OR appeared in 2025 (ODA & ODF, 2025; ODA & ODF, n.d.).

Oregon ash trees range from western Washington and Oregon to central California and are most commonly found in well-drained soils in riparian areas or wet clay soils in savanna-wet prairie sites wetlands (Kral & Shaw, 2023), though there are no published estimates of Oregon ash land cover. Oregon ash functions as both an early and late colonizing species and can occur as a smaller understory tree or as the major overstory canopy (Prive, 2016). Like other ash species, Oregon ash trees are also often planted in urban landscapes throughout the Pacific Northwest. Oregon ash is a culturally important tree to Indigenous nations across the West Coast, who have used its wood for fuel and various tools, as well as its bark and roots for medicinal purposes (Chestnut, 1906; Gunther, 1973). The IUCN Red List assessment categorizes Oregon ash as "Near Threatened" due to urban expansion, logging, and vulnerability to EAB (Oldfield &

Westwood, 2017). This assessment has not been updated since the introduction of EAB to Oregon ash's range and therefore the current IUCN status is likely more advanced.

As EAB spreads throughout the Pacific Northwest, Oregon ash mortality due to EAB may have different negative effects depending on the habitat niche. Riparian areas might be relatively more resilient to ash loss due to higher forest canopy diversity, though reduced ash canopy is still expected to have significant impacts on stream temperatures and therefore species like salmon and other aquatic organisms that need cool water (Maze et al. 2024). Conversely, the savanna-wet prairie sites Oregon ash inhabits have lower species diversity and are often dominated by Oregon ash with small amounts of Oregon white oaks (Prive, 2016). Oregon ash overstory in these places prevents exotic species like Himalayan blackberry (*Rubus armeniacus*) from proliferating, but ash mortality may cause these species to expand in its place (Prive, 2016).

Management efforts in outbreaks across the Midwest and East Coast have largely been unsuccessful in eradicating EAB, with varying degrees of limited success in slowing spread (Sun et al., 2024). Early detection of EAB has proved difficult, relying on individual tree health assessments and sticky traps that catch live adult EAB beetles. Chemical control via trunk pesticide injection is effective but costly and not practical for landscape scale eradication (Herms et al., 2019). Biological control using hymenopteran and coleopteran natural enemies has been moderately successful, but parasitoids exhibit much lower efficiency in the introduced range compared to their native ranges (Duan et al., 2018). Selective breeding of resistant native ash or hybridization with ash trees from EAB's native range is a slow process and still an active area of research; while some successful breeding has occurred, resistance of these trees is still unknown (Kelly et al. 2020; Koch et al., 2012). More research is needed to determine the genetic source of ash resistance to EAB. The mixed success of these previous management approaches has pushed land managers in the Pacific Northwest to shift resources from eradication to slowing spread and preparing for inevitable EAB arrival. Current management efforts in Oregon are aimed at monitoring using sticky traps and surveys, slowing the spread of EAB by promoting "Don't Move the Firewood" campaigns, and preparing for EAB arrival and subsequent damage by studying and planting potential replacement species (Shaw et al., 2025; Sun et al. 2024).

There is a need for more information to guide management and restoration as local land managers and owners prepare for high levels of ash mortality due to EAB. Forest landscape modelling is a powerful tool that can project current forest conditions into the future so that

researchers and land managers can anticipate forest changes. This tool, particularly one called LANDIS-II (Scheller et al., 2007), was applied before by Olson et al. (2021) to an EAB outbreak in northern Wisconsin. So far, the only modelling of EAB in the Pacific Northwest was recently done by Zhu et al. (2025) where they modeled EAB habitat suitability. Zhu et al.'s results predict EAB in locations where there are no ash, because they used deciduous tree presence/absence as the vegetation indicator. LANDIS-II is an improvement for a number of reasons, particularly because it enables targeted analysis of specific tree species and age classes, leading to a more precise examination of forest pest dynamics while also providing a broader understanding of landscape-level forest change through time.

To address this research gap, this study utilized a spatially explicit, process-based forest landscape model, LANDIS-II, to project the future impact of the emerald ash borer on Oregon ash. Specifically, this research aimed to ask: (1) how will emerald ash borer spread across Oregon and Washington over the next 25 years? And (2) how might plant community successional trajectories be altered in areas affected by the emerald ash borer?

METHODS

Study Area

The study area for this research encompasses approximately 3 million hectares in the Puget Lowland and Willamette Valley regions of western Washington and western Oregon (Figure 1). The extent of the study area was determined by considering 1) the highest density and range of Oregon ash and 2) the extent to which EAB might reasonably spread by 2050. The specific boundaries of the study area are the Environmental Protection Agency's Level IV ecoregions that fit these two considerations (Omernik, 2014). Elevation in this area ranges from sea level to 955 meters (Hollister, 2023). The climate is characterized as Mediterranean, experiencing warm, dry summers and cool, wet winters. Average precipitation is 142.1 cm per year, average minimum temperature is 5.8°C, and average maximum temperature is 16.3°C (PRISM Climate Group, 2024). A range of land cover types occupy this landscape with 38% of the land area designated as forest, 26% as agriculture, and 21% is developed (U.S. Geological Survey (USGS), 2024). The remaining 15% includes land cover types such as open water, wetlands, and shrublands.

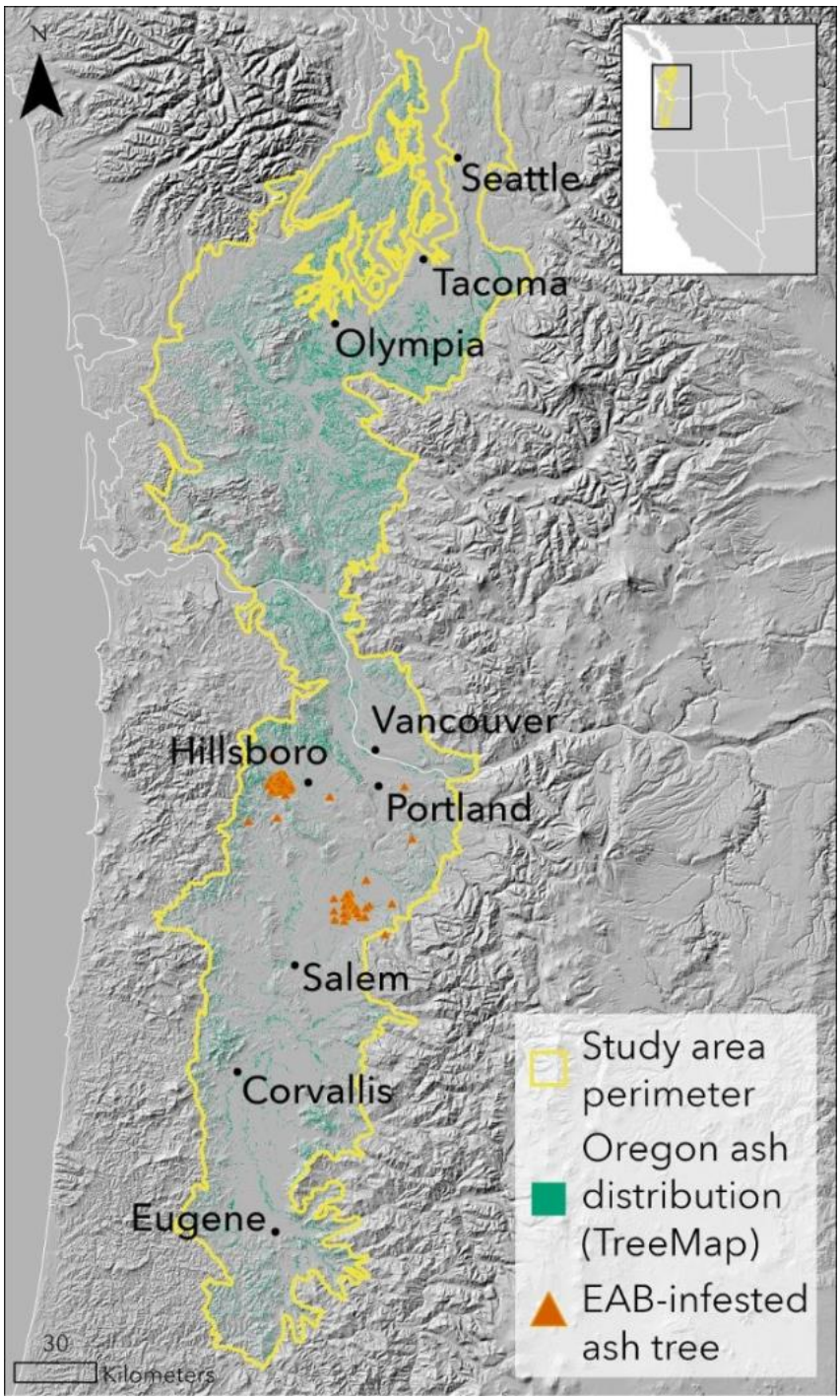


Figure 1. *Map of project study area in the Pacific Northwest. Yellow line is the study landscape perimeter, green cells represent Oregon ash distribution as determined by the US Forest Service TreeMap product, and orange triangles represent surveyed ash trees infested with EAB (Houtman et al., 2025; ODA & ODF, 2025)*

Forests across this region can be categorized into three main forest types (Franklin & Dyrness, 1973; Van Pelt, 2007). Western hemlock forests are the most prominent type,

consisting primarily of western hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*), and western redcedar (*Thuja plicata*). These forests are located throughout western Washington and in the lowlands of Oregon's Coast Range and Western Cascades. Oak woodlands with Oregon white oak (*Quercus garryana*), bigleaf maple (*Acer macrophylum*), Douglas fir, and Pacific madrone (*Arbutus menziesii*) are found in the southern Puget Sound and Willamette Valley regions. This forest type also includes Oregon ash (*Fraxinus latifolia*) and Oregon white oak wetland forests. Finally, conifer forests found in the foothills of the interior valleys of this area are composed of Douglas fir, grand fir (*Abies grandis*), bigleaf maple, and in the more southerly areas, ponderosa pine (*Pinus ponderosa*).

Model framework

The forest landscape model LANDIS-II (version 8) was used to project the spread of Emerald ash borer, subsequent Oregon ash mortality and post-EAB successional trajectories. LANDIS-II is a process-based, spatially interactive forest landscape model that simulates forest dynamics and succession over large spatial and temporal scales (Scheller et al., 2007). The model is raster-based and begins with initial maps of species-age cohorts, advancing the cohorts through time based on species life history strategies, abiotic factors, competition, and, optionally, disturbances. LANDIS-II is open-source and customizable, enabling users to employ different extensions that suit the research objectives and landscape. These extensions include forest processes such as insect disturbance, fire, forest management, wind disturbance, and habitat suitability. LANDIS-II has previously been applied to Pacific Northwest forests (Mast et al., 2025; Laflower et al., 2016; Creutzburg et al., 2016) and emerald ash borer-threatened forests in Wisconsin (Gustafson et al., 2025; Olson et al., 2021), though never to the emerald ash borer invasion in the Pacific Northwest.

The Net Ecosystem Carbon and Nitrogen Succession (NECN) extension (version 8.1.3) was used to simulate forest successional processes at a monthly timestep across the landscape (Scheller et al., 2011; Scheller et al., 2012). In NECN, succession is an emergent property of abiotic factors and each species' life history strategies. Tree growth occurs as a function of species-specific tolerances to water availability, light availability (as a function of leaf area index [LAI]), temperature, growing space capacity, and nitrogen availability. Cohorts advance in age until reaching their longevity, unless mortality is triggered by disturbance. NECN uses

precipitation and soil inputs to calculate water balance components like runoff and evapotranspiration throughout the system, which impact soil water availability for plant growth. Reproduction can occur in two ways at each NECN timestep: 1) after a disturbance via planting, serotiny, or resprouting, and 2) via dispersal into a site. Provided there is sufficient LAI at a site, establishment is limited by LAI, growing degree days, drought tolerance, and minimum January temperature. This extension also tracks carbon and nitrogen cycling across live biomass, soil, litter, and dead woody debris pools (Scheller et al., 2011; Lucash et al., 2014).

The Climate Biological Disturbance Agent (BDA) extension (version 5.1.1) was used to simulate EAB-related ash mortality across the landscape (Sturtevant et al., 2004). BDA models the effects of insect disturbance on forest ecosystems as a function of pest life history, pest outbreak patterns, host distribution, and other factors. The extension calculates the probability of outbreak at each cell to simulate patterns of spread and produce maps showing disturbance severity. After the initial epicenters are established in year 1, subsequent outbreaks may occur, optionally, through (1) deterministic cell-to-cell spread governed by dispersal parameters or (2) stochastic establishment of new epicenters determined by seed-epicenter settings. Disturbance is species- and cohort-specific, as determined by the host input parameters.

In previous versions of BDA (e.g. Olson et al., 2021), the location of initial outbreak epicenters was randomized across the landscape. A recent model improvement allows the user to provide a map describing known locations of disturbance agent presence. The optional raster map contains values 0 to 1 describing the probability of outbreak at each individual cell for the first timestep. This update now allows users to select where biological disturbance agents initiate and spread from, providing the opportunity to simulate ongoing or hypothetical outbreaks in specific locations.

Parameterization

Climate and vegetation inputs

LANDIS-II and its extensions require raster and tabular inputs that describe the climate, soils, and vegetation of the landscape. A 90-meter resolution was selected for this project to balance model computational efficiency and result precision across the large landscape. All raster inputs were processed using the terra package in RStudio (Hijmans, 2025; Posit team, 2025).

In LANDIS-II, climate data is averaged across climate regions so that conditions are homogenous within each region. Climate regions were created by aggregating 6km^2 CMIP6 General Circulation Model (GCM) grids from LOCA2 to 18 km^2 (Pierce et al., 2023). This resulted in a total of 150 climate regions across the landscape (Fig. S1). To prevent establishment in unsuitable habitats (e.g. urban areas, water, barren rock), these areas were masked from the climate region map using National Land Cover Database (NLCD) land cover data (USGS, 2024). Historic daily precipitation, minimum temperature, and maximum temperature data from 1979 to 2020 were downloaded from gridMET (Abatzoglou, 2013), aggregated to monthly values, and assigned to each climate region (Fig. S2).

Initial species composition and biomass was derived from TreeMap 2022, a U.S. Forest Service map product of continuous imputed forest structure that uses Forest Inventory and Analysis (FIA) plot data (Houtman et al., 2025; U.S. Forest Service, 2023). Tree species present in more than 1% of TreeMap sites were included in the list of tree species to be modeled; non-native species were excluded. Important Oregon ash shrub associates, native and non-native, present in more than 20% of TreeMap sites were also included. This resulted in a total of 20 tree species and 8 shrub species (Table 1).

Table 1. *Modelled species list.*

Common Name	Scientific Name	Species type
Bigleaf maple	*Acer macrophylum*	Tree
Bitter cherry	*Prunus emarginata*	Tree
Black cottonwood	*Populus balsamifera* ssp. *trichocarpa*	Tree
California black oak	*Quercus kelloggii*	Tree
Canyon live oak	*Quercus chrysolepsis*	Tree
Douglas-fir	*Pseudotsuga menziesii*	Tree
Grand fir	*Abies grandis*	Tree
Incense cedar	*Calocedrus decurrens*	Tree
Lodgepole pine	*Pinus contorta*	Tree
Oregon ash	*Fraxinus latifolia*	Tree
Oregon white oak	*Quercus garryana*	Tree
Pacific madrone	*Arbutus menziesii*	Tree
Pacific silver fir	*Abies amabilis*	Tree
Pacific yew	*Taxus brevifolia*	Tree
Ponderosa pine	*Pinus ponderosa*	Tree
Red alder	*Alnus rubra*	Tree
Sitka spruce	*Picea sitchensis*	Tree
Western hemlock	*Tsuga heterophylla*	Tree

Western redcedar	*Thuja plicata*	Tree
Western white pine	*Pinus monticola*	Tree
Beaked hazelnut	*Corylus cornuta*	Shrub
California blackberry	*Rubus ursinus*	Shrub
California hazelnut	*Corylus cornuta var. californica*	Shrub
Common snowberry	*Symphoricarpos albus*	Shrub
Himalayan blackberry	*Rubus armeniacas*	Shrub
Poison oak	*Toxicodendron pubescens*	Shrub
Rose spirea	*Spireaea douglasii*	Shrub
Salmonberry	*Rubus spectabilis*	Shrub

The initial communities map requires an associated table that describes the species-age cohorts present in each cell and their associated wood and leaf biomass; this information was also derived from FIA plot data (U.S. Forest Service, 2023). Tree ages are available for some but not all trees in FIA. To estimate missing tree ages, a linear regression model with tree height and DBH was fitted for each species. Shrub ages were determined through a rule-based method incorporating stand age, longevity, shade tolerance, and percent cover data. Shrub data in FIA is provided as percent cover, so biomass was calculated using a shrub-specific allometric equation from the literature (Chojnacky & Milton, 2008).

NECN

Species life history traits such as longevity, sexual maturity, seed dispersal, and vegetative reproduction probability were gathered from databases including TRY (Kattge et al., 2020), Fire Effects Information System (FEIS), and Silvics of North America (Burns & Honkala, 1990), as well as past LANDIS-II studies from the region (Mast et al., 2025). NECN requires additional species parameters regarding carbon and nitrogen cycling, leaf area index, and decomposition, which were also derived from the literature (Table S1). Soil moisture and temperature growth response curves were fitted for functional groups (hardwoods, dry conifers, and wet conifers) by combining summarized monthly temperatures from GridMET (Abatzoglou, 2013) with MODIS/Terra Gross Primary Productivity (GPP) dataset (Running et al., 2015) at 20 different point locations that contain species representing the above functional groups throughout their range (Fig. S3).

NECN uses a suite of soils data related to soil texture, soil hydrology, and soil organic matter. Soil data was primarily derived from Soil Landscapes of the United States (SOLUS), a national set of spatially complete soil maps at 100m resolution (Soil Survey Staff, 2024).

SoilGrids, an international source of 250m resolution soil data, was used to fill any gaps in SOLUS data (Poggio et al., 2021). Dead coarse root and dead woody debris maps were derived from a US Forest Service carbon stocks dataset (Wilson et al., 2013).

BDA

EAB survey data from the greater Portland, OR area was used to create the initial EAB epicenter map (ODA & ODF, 2025). This point data on EAB-infested ash trees was used to create a landscape raster map of 1s and 0s; 1s for cell overlapping with EAB-infested ash trees and 0s elsewhere. Because tree mapping is less accurate in urban areas, especially where non-native ash trees may be used for landscaping, only 6 out of the 186 infested cells overlapped with ash presence in the initial communities map. To resolve this discrepancy, the infested ash tree cells were buffered by 500 meters to capture additional ash trees on the landscape. BDA allows the user to select the number of initial epicenters. The maximum number of cells with EAB infested trees (160) was selected so that EAB could spread from all infested cells.

BDA input parameters which describe the disturbance agent outbreak and dispersal traits, as well as the host species' susceptibility were derived from known pest-host relationships (Knight et al., 2013; Haack et al., 2015; Poland et al., 2015). Three dispersal rates were selected to represent the range of estimates for EAB annual dispersal capabilities: 0.4 km/year, 5.2 km/year, and 10 km/year (Mercader et al., 2016; Taylor et al., 2010). To account for long distance dispersal events (i.e. movement of EAB-infested wood products by humans), the seed epicenter function in BDA was utilized. The maximum number of seed epicenters input was determined by calculating an average of observed new epicenters in the Portland, OR area outbreak from 2023 to 2024 and 2024 to 2025 (ODA & ODF, 2025).

Calibration

Single cell simulations were used to adjust species level parameters while full landscape simulations were used for landscape scale parameters. First, wet and dry atmospheric nitrogen deposition was calibrated to empirical values (Fig. S4; National Atmospheric Deposition Program (NRSP-3), 2022). Species-specific fractions of annual net primary productivity (ANPP) to be allocated to leaves, fine roots, and coarse roots were calibrated to mean foliar biomass ratios and fine/coarse root biomass ratios from FIA (Table S1; U.S. Forest Service, 2023).

Growth was calibrated by comparing age-biomass relationships of simulation results to FIA data (U.S. Forest Service, 2023). MaxBiomass and MaxANPP were adjusted until simulated age-biomass relationships aligned with empirical data (Fig. S5, Fig. S6). Leaf area index (LAI) values over time for each species were calibrated by adjusting maxLAI and KLAI until LAI approached maxLAI at ages comparable to empirical sources. Regeneration was calibrated by comparing the initial number of young trees per species on the landscape to the number of young trees reproduced over time in LANDIS-II to ensure they were consistent over time (Fig. S7, Fig. S8).

Analysis

A full factorial design of EAB dispersal rates was used and simulations were run from 2022 until 2050 at a 1-year timestep. Three dispersal rates representing low (0.4km/year), medium (5.2 km/year), and high (10 km/year) estimates of EAB dispersal found in the literature were tested (Mercader et al., 2016; Taylor et al., 2010). These dispersal rates were simulated both with and without long-range anthropogenic EAB dispersal for a total of 15 simulations (3 dispersal rates * 2 human dispersal scenarios * 3 replicates).

Data wrangling, analysis, and visualization were done using R in R Studio (R Core Team, 2024; Posit team, 2025).

RESULTS

Emerald ash borer spread across the landscape was limited by ash distribution in the low dispersal rate scenarios, but not in the medium or high dispersal rate scenarios (Fig. 2). The low EAB dispersal rate restricted EAB to the greater Portland, OR and Woodburn, OR areas through 2050. At the low dispersal rates (0.4 km/year), spread was limited by ash availability, with simulated dispersal rates as low as 0.2 km/year. In both the medium and high dispersal rate scenarios, spread was not limited by ash availability and simulated dispersal rates were similar to the input (5.2 km/year for medium, 10 km/year for high). The medium dispersal rate allowed EAB to reach south of Olympia, WA in the north and Eugene, OR in the south by 2050., while the high dispersal rate allowed EAB to reach the landscape's southern border by ~2040 and the northern border by ~2050.

Adding human-aided transport of wood material (long-distance dispersal, LDD) significantly altered the outbreak's spatial pattern and increased the rate of EAB spread across the landscape (Fig. 2). In the low dispersal scenarios, LDD caused a patchy spread pattern of epicenters across the entire landscape, reaching the northern and southern landscape extents by 2030 at an simulated dispersal rate 20x greater than the input rate (7.9 vs 0.4 km/year). In both the medium and high dispersal rate scenarios, LDD caused continuous EAB spread throughout the extent of the landscape by ~2035 and ~2025, respectively. At the medium dispersal rate, LDD allowed EAB to spread continuously throughout the landscape at an simulated rate 4x greater than the input rate (22.4 vs 5.2 km/year). In the high dispersal rate scenario, EAB spread across the entire landscape well before 2050, with LDD increasing the spread rate three-fold (30.5 vs 10 km/year).

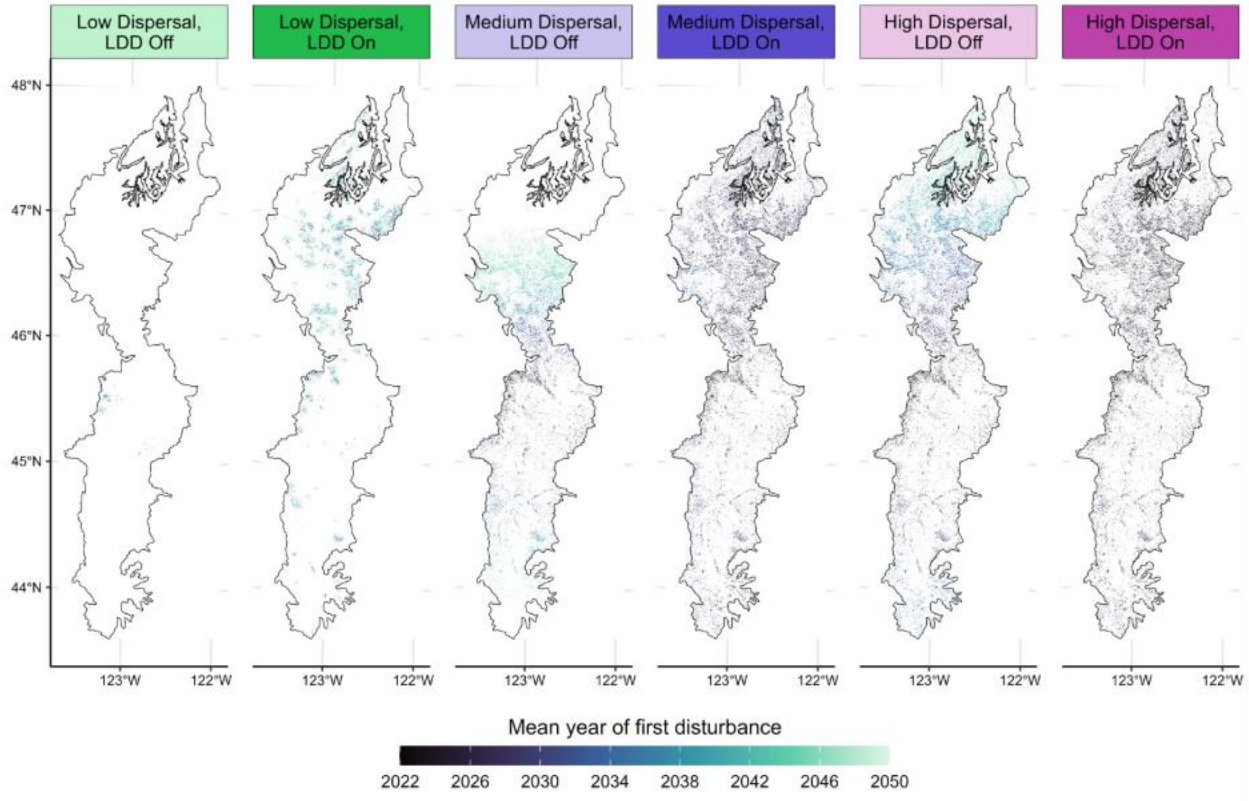


Figure 2. *Average year of first EAB disturbance across three replicates by dispersal scenario. Dispersal scenarios reflect estimates of EAB dispersal in the literature, and long-distance dispersal (LDD) indicates simulated human transport (LDD On) or no human transport (LDD Off).*

EAB caused declines in the biomass of Oregon ash across all dispersal scenarios; ash was eliminated from the landscape by 2050 in three of six scenarios (Fig. 3). Scenarios without human-aided transport of wood material (LDD off) experienced linear decreases in Oregon ash biomass. By 2050, Oregon ash declined by 21.5% under the low dispersal scenario and by 78.5% under the medium dispersal scenario. Oregon ash was effectively eliminated from the landscape by 2045 in the high dispersal scenario. In scenarios with LDD, Oregon ash biomass declined linearly in the low dispersal scenario, whereas the medium and high dispersal rate scenarios caused exponential decreases in biomass. In the low dispersal rate scenario, adding LDD led to a 36.2% decrease in Oregon ash biomass by 2050. Both the medium and high dispersal rate scenarios with LDD led Oregon ash to be effectively eliminated from the landscape by 2036 and 2030, respectively.

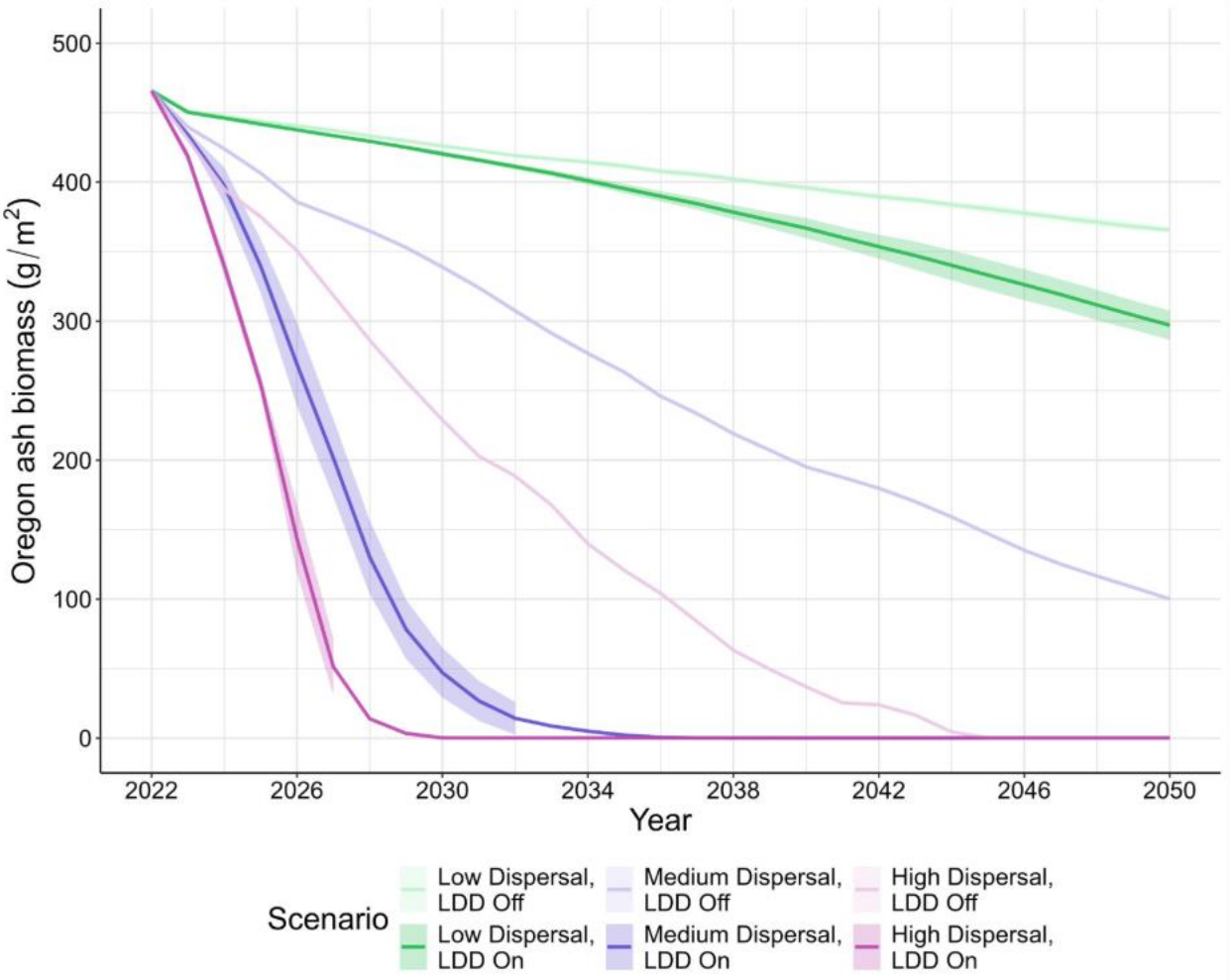


Figure 3. *Simulated Oregon ash biomass density over time averaged across three replicates by dispersal scenario. Dispersal scenarios reflect estimates of EAB dispersal in the literature and long-distance dispersal (LDD) indicates simulated human transport (LDD On) or no human transport (LDD Off).*

The dispersal scenarios produced different spatial patterns of change in Oregon ash biomass (Fig. 4). In scenarios without LDD, the low dispersal rate led to minimal changes in Oregon ash biomass across most of the landscape by 2050. Decreases in biomass were observed surrounding the current outbreak areas near Forest Grove and Woodburn, OR, while increases in biomass occurred at the northern extent of the landscape around the Kitsap Peninsula. The medium dispersal rate caused more widespread decreases in biomass across the middle section of the landscape, with minimal changes in biomass at the northern and southern ends of the landscape, though ash increased in biomass in the Kitsap Peninsula. The high dispersal rate showed complete elimination of Oregon ash biomass throughout the entire landscape by 2050. In scenarios with LDD, the low dispersal rate led to patchy areas of complete elimination of Oregon

ash biomass with some areas of no significant change in biomass. The Kitsap Peninsula remained the only area where Oregon ash biomass increased. Both the medium and high dispersal rate scenarios with LDD resulted in the complete elimination of Oregon ash biomass across the landscape by 2050.

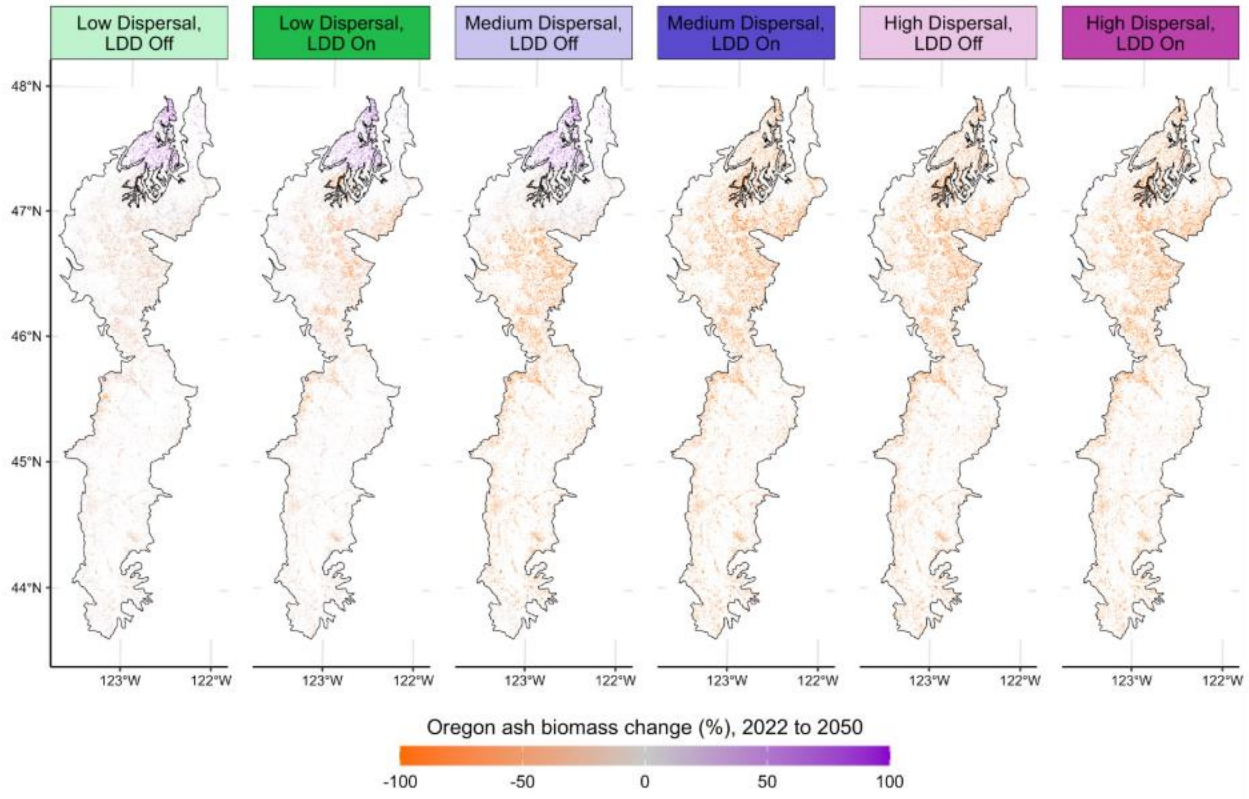


Figure 4. *Simulated change in Oregon ash biomass density from 2022 to 2050 averaged across two replicates by dispersal scenario. Dispersal scenarios reflect estimates of EAB dispersal in the literature and long-distance dispersal (LDD) indicates simulated human transport (LDD On) or no human transport (LDD Off).*

In sites across the landscape where Oregon ash was the dominant species and EAB disturbance occurred, the decline in Oregon ash biomass was most frequently associated with an increase in the biomass of western redcedar (Fig. 5). On average, western red cedar biomass increased by 19.2% across all regions. Other prominent species co-occurring with Oregon ash were black cottonwood, red alder, and Douglas fir. Slight regional differences in successional trajectories occurred with these species. Black cottonwood experienced declines in biomass across all regions, with smaller decreases (4.7%) in southern regions (greater Corvallis and Eugene areas) compared to greater decreases (27%) in northern regions (greater Portland,

Centralia, and Tacoma areas). Red alder, which was more prevalent in central and southern regions of the landscape (greater Portland, Corvallis and Eugene areas), experienced a 39% decline in biomass after EAB outbreak. Douglas fir also experienced decreases in biomass (90%) in the southern regions (greater Corvallis and Eugene areas), but increased in northern regions (greater Portland, Centralia, and Tacoma areas) by an average of 244%.

Regional differences in shrub successional trajectories were even more apparent than those of trees in ash-dominant sites following EAB disturbance (Fig. 5). California and beaked hazelnut were two prominent shrub species across the landscape, with California hazelnut dominating in northern regions (greater Centralia and Tacoma areas) and beaked hazelnut dominating in southern regions (greater Portland, Corvallis, and Eugene areas). In some regions (greater Eugene and Tacoma areas), California hazelnut increased by 2.8-fold after invasion, while in other regions (greater Centralia, Portland, and Corvallis areas) it decreased by an average of 9%. Similarly, Beaked hazelnut increased by 13-fold in some regions (greater Tacoma and Corvallis areas) while in other regions (greater Centralia, Portland, and Eugene areas), it decreased by only 19%. Common snowberry and Rose Spirea were initially prevalent in the greater Portland and Corvallis areas, but over time were eliminated from the landscape. Poison oak was initially most prevalent in the greater Eugene area, but biomass of poison oak decreased by 21% over time since disturbance. Himalayan blackberry increased by 5.6-fold in the greater Eugene and Tacoma areas but decreased by 98% in the greater Centralia, Portland, and Corvallis areas. Salmonberry exhibited both increases and decreases depending on the region; in the Tacoma and Eugene areas it increased by an average of 225% but in Centralia, Portland, and Corvallis, it decreased by an average of 85%.

Despite increases in some tree and shrub biomass after EAB outbreak, none of the landscapes recovered enough biomass by the end of the modelled time period to match pre-Oregon ash mortality levels. Even in areas where EAB disturbance occurred early in the simulations, allowing more time for succession and recovery, biomass did not recover. Average total tree and shrub biomass across regions was 10,754 g/m^2 before EAB invasion, while the average final biomass was 2,819 g/m^2.

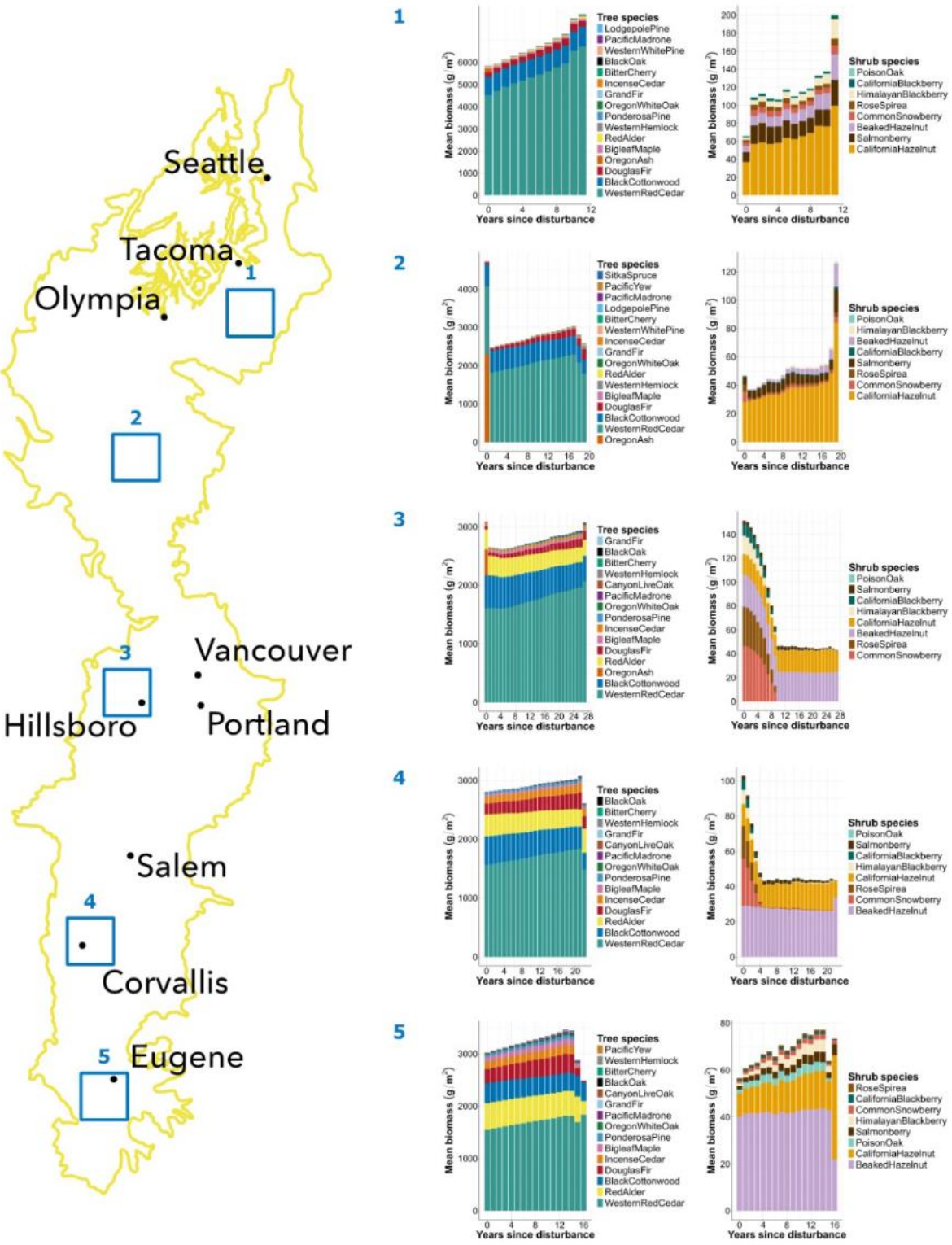


Figure 5. *Mean biomass of tree and shrub species in Oregon ash dominant sites across five subregions in the landscape, averaged across years since EAB disturbance.*

DISCUSSION

The research described here expands upon previous EAB modelling efforts in the upper Midwest and Pacific Northwest. In previous studies using LANDIS-II BDA, initial outbreaks occurred randomly across a landscape and did not correspond with empirical data of initial outbreaks (e.g. Olson et al., 2021). Now, with the model changes to include an initial epicenter map, it is possible to simulate spread from existing epicenters. This allows for the ability to model real outbreaks from their current epicenters, giving land managers and owners the insight they need to prepare for future spread. This work also improves on other EAB modelling work done in the Pacific Northwest. Zhu et al. (2025) created a habitat suitability model for EAB in the Pacific Northwest and used this to predict future spread from previous outbreak epicenters. While Zhu et al. used deciduous tree presence/absence, this study used ash distribution with biomass and age data from TreeMap (Houtman et al., 2025). This is an important improvement that allows for more precise modelling, targeting the actual host species rather than a large group of trees, most of which are unsuitable hosts for EAB. To simulate EAB spread, Zhu et al. used short and long-distance dispersal rates of 20km and 47km that both account for natural EAB spread and human-aided transport of wood materials. In this study, a range of EAB dispersal rates were tested but human-aided dispersal events were accounted for separately, enabling the teasing apart of these two different types of dispersal.

The results indicate that a threshold response for EAB spread exists between the low and medium dispersal rates. Oregon ash distribution is not continuous enough near initial epicenters to support EAB spread at a low dispersal rate. Therefore, if this current outbreak near Portland, OR can be contained to its current distribution and the EAB dispersal rate is low, it may be possible to prevent further spread and Oregon ash mortality. However, at the medium and high simulated dispersal rates, EAB was able to spread regardless of ash continuity. There are currently no studies that investigate the impact of ash continuity on EAB dispersal rate, likely because previous EAB outbreaks occurred in the Midwest and eastern United States where ash forests are relatively more continuous. Past studies only focused on tree density but not extent or spatial patterns; EAB invasion risk was previously found to be positively associated with ash density in the eastern United States (Ward et al., 2020).

Limiting human-aided transport of Oregon ash wood materials could greatly impact the future outcomes of the current EAB outbreak in the Pacific Northwest. Adding human-aided

long distance dispersal events to simulations accelerated EAB spread and resulting Oregon ash mortality, especially at low and medium EAB dispersal rates. High dispersal rates led to landscape-level elimination of ash before 2050 regardless of human-aided dispersal, however it did accelerate the rate of spread and resulting mortality. Efforts like The Nature Conservancy's "Don't Move Firewood" campaigns and other public outreach have the potential to greatly influence the outcomes of this outbreak by slowing the human-aided spread across the landscape (Don't Move Firewood, n.d.). A recent modelling study found quarantine efforts when combined with biocontrol, even at less than 100% quarantine efficiency, were found to be the most effective (Hudgins et al., 2024). Further research on the effectiveness of education campaigns contributing to the level of quarantine efficiency is needed.

Zhu et al. found that EAB could reach Washington in 2 years, spread throughout western Oregon in 15 years, and reach California by 20 years, although the survey year used to initialize the model is unclear. The results presented here indicate that EAB's expected timing will vary depending on the dispersal rate, however, Zhu et al.'s findings most closely align with this study's medium dispersal, LDD off scenario. This study's low dispersal rate led to slower spread than Zhu et al.'s and the high dispersal rate led to faster. Significant differences in the resulting spatial patterns exist, with Zhu et al.'s model showing EAB spread to suitable areas that lie outside of the range of Oregon ash.

The results indicate that western redcedar is the species most likely to replace Oregon ash in areas where Oregon ash currently dominates. However, this represents an associated increase in western redcedar biomass rather than a direct Oregon ash replacement. The expectation that common ash-associated tree species like Oregon white oak, black cottonwood, red alder, and bigleaf maple would increase in place of Oregon ash was not realized in these simulations. Similarly, the expectation that Oregon ash-associated shrub species would increase did not occur. This is likely due to the source used for initial vegetation conditions, TreeMap, which did not show these associations to the extent that they are observed in the field. These initial vegetation conditions drove the successional trajectories observed in the model.

Oregon ash dominant sites never recovered to pre-Oregon ash mortality biomass amounts, suggesting that the loss of ash will likely cause a decrease carbon storage capacity of these sites. Additionally, the loss of forested canopy cover is likely to increase stream temperatures in riparian areas, which can be detrimental to aquatic wildlife (Maze et al., 2024).

Counter to anticipated results, these results do not show significant amounts of invasive species like Himalayan blackberry at the landscape scale (Prive, 2016). Still, restoration after Oregon ash mortality is likely to be important for ensuring ecosystem recovery given this loss of biomass. The significant loss of forest canopy indicated by these results is likely to constrain restoration options, as Hull (2024) recommends managing for non-forested wetlands post-EAB disturbance given the limited options for suitable replacement tree species and an expected rise in the water table.

Model improvements that now allow the use of an initial epicenter map, a significant improvement from past model capabilities, still cannot always be calibrated to the multiple years of outbreak data available. Replicating the spatial pattern of EAB outbreaks, particularly human-caused outbreaks is difficult due to stochasticity in the model. LANDIS-II BDA simulates random epicenters, defined by ash availability and dispersal distances, but the location where human transport of woody material might occur is unpredictable. In this study, outbreak data from 2022-2025 was combined to create an initial outbreak map that would match empirical observations. This translates to a slight mismatch between the initial vegetation map from 2022, and outbreak data combined from 2022 to 2025. Additionally, BDA cannot simulate the lag in ash mortality that takes place after infestation with EAB. EAB is a fast-acting tree mortality agent compared to others, but it still takes ash trees around three to five years to die after EAB infestation (Poland & McCullough, 2006).

Although TreeMap (Houtman et al., 2025) is the best source of initial vegetation conditions for this research, it has limited accuracy in urban areas, which, if improved, could strengthen predictions near current outbreak epicenters. Many current EAB epicenters did not overlap with TreeMap's ash distribution, indicating TreeMap either underestimates urban Oregon ash or excludes it due to urban land cover classification. Additionally, there is little representation of Oregon ash in the FIA plots imputed across landscape in TreeMap product; only 17 unique plots contain Oregon ash (Walker, unpublished data). The recently published Willamette Valley Hardwoods Map (McKenzie et al., 2026) could help resolve this issue, but it is only available for the Willamette Valley; EAB is likely to spread well beyond the Willamette Valley in the coming years. Also, the map only provides presence/absence data for hardwood species, but woody biomass is a required input for LANDIS-II.

An additional limitation in this research is the lack of literature on Oregon ash and shrub species. Limited literature can be found on Oregon ash ecology and physiology, likely because it is not a prominent timber species. The work here relied on additional information sources such as Master's theses (e.g. Prive, 2016; Hull, 2024) and extension service reports (e.g. Kral & Shaw, 2023; Shaw, 2025), highlighting the need for more research on Oregon ash as a species. Improving understanding of this species as a whole will better equip land managers to protect it. Additionally, TreeMap is not an ideal data source for parameterizing shrub data. FIA provides percent shrub cover, which must be converted to biomass and age for use in LANDIS-II. When scaled to the landscape level, this may misrepresent different shrub species' abundance. Himalayan blackberry and poison oak had surprisingly low biomass in ash-dominant sites, despite EAB experts suggesting they co-occur with ash and would expand after ash loss (Shaw, per comm). The results here did not support this hypothesis. It is possible that FIA does not sample enough near riparian areas and therefore misrepresents this Oregon ash habitat niche.

Improved estimates of EAB dispersal capabilities would enhance modelling accuracy, allowing for more actionable predictions for land managers. The dispersal rates used here represent values derived from laboratory experiments (Taylor et al., 2010) or observational studies in the Midwest (Mercader et al., 2016). While flight mill research in the laboratory is able to test the physiological limits of the emerald ash borer, they may not accurately reflect on the ground dynamics of the insect when faced with host availability and other constraints. Additionally, observed dispersal rate data from the Midwest might not translate to the Pacific Northwest due to different ash connectivity and climate dynamics. Given there are only three years of EAB field observations in the greater Portland, OR area outbreak, in addition to uncertainty regarding when and where the outbreak began, it is difficult to calculate a dispersal rate specific to this early-stage outbreak. EAB spread can vary depending on proximity to previous epicenters, human population density, and ash density, so it is likely that EAB spread rates will vary across space and time (Ward et al., 2020). EAB is unlikely to face climatic constraints in the Pacific Northwest, given that the beetle is primarily limited by extreme cold temperatures (DeSantis et al., 2013). Additionally, it is difficult to separate natural EAB spread from human-aided spread. While longer distance dispersal events such as from Colorado to Oregon or the jump from the initial Pacific Northwest epicenter in Forest Grove, OR in 2022 to Woodburn, OR in 2023 are easier to attribute to human-aided dispersal, it is more difficult to

determine at smaller distances. Understanding a more realistic range of possibilities may help to improve accuracy of modelling efforts.

Future work might expand upon this research by updating model parameters as more is learned about the outbreak patterns of EAB and Oregon ash distribution in the Pacific Northwest. Additionally, adding climate change scenarios might provide further insights into how successional trajectories might change with increasing temperature or altered precipitation patterns. Future modeling studies that use the Biomass Harvest extension in LANDIS-II, which allows for forestry management actions to be implemented in forest landscapes, including tree harvesting, thinning, and planting (Gustafson et al., 2000), may be helpful for simulating different restoration strategies. This extension could be used to simulate replanting of different candidate species to determine which species best maintain the ecosystem services currently provided by ash, such as leaf area and carbon storage.

CONCLUSIONS

It may be possible to contain the current EAB outbreak in the Pacific Northwest to the greater Portland, OR area if human-aided transport is kept to a minimum and EAB disperses at a low rate. There is a range of future possible outcomes of EAB spread and resulting Oregon ash mortality that depend on the EAB dispersal rate and extent of human-aided transport of infested wood products that take place, however, landscape-level elimination of Oregon ash by 2050 is likely. Western redcedar can be expected to increase in biomass after EAB-related Oregon ash mortality, however, pre-disturbance biomass is not recovered in any regions across the landscape by 2050. Improved EAB dispersal rate estimates, Oregon ash mapping, and other research on Oregon ash's ecological role and physiology are needed to improve these model predictions of EAB spread and Oregon ash mortality.

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