FINAL REPORT

Modeling the Carbon Implications of Ecologically Based Forest Management

SERDP Project RC-2118

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14. ABSTRACT
Increasing treatment intensity resulted in decreasing total ecosystem carbon (TEC). Treatments that included thinning, however, increased net ecosystem carbon balance, making thinned forests a larger sink for carbon. This finding is a result of decreased competition for resources resulting from thinning. At Fort Benning actively restoring longleaf pine forest (LLP) decreased TEC by approximately 22% compared to the control (broadleaved forest). LLP restoration, however, increased red-cockaded woodpecker habitat such that approximately 90% of the upland forest area (e.g. historically LLP forest) was viable habitat by the end of the simulation period. At Camp Navajo thinning and prescribed burning treatments decreased TEC relative to the control in the absence of wildfire, but increased TEC relative to the control when wildfire was simulated. Simulations also demonstrated that thinning and burning decreases the risk of habitat loss for the Mexican spotted owl. At Joint Base Lewis McChord treatments that included thinning resulted in the lowest TEC, but the highest net ecosystem carbon balance. In addition, thin-only and thin and burn treatments significantly increased the probability of Oregon white oak presence by the end of the simulation period, an important food source for western gray squirrel during mast years.

15. SUBJECT TERMS
Forest carbon, wildfire, management, thinning, prescribed burning

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List of Acronyms

BA – Basal Area
C – Carbon
CN – Camp Navajo
CV – Coefficient of Variation
DBH – Diameter at Breast Height (1.37 m)
DoD – Department of Defense
FB – Fort Benning
FVS – Forest Vegetation Simulator
ICO – Individual trees, Clumps of trees, Openings
JBLM – Joint Base Lewis McChord
MSO – Mexican Spotted Owl
NCDC - National Climatic Data Center
NEE – Net Ecosystem Exchange
NEP – Net Ecosystem Production
NPP – Net Primary Productivity
PDSI – Palmer Drought Severity Index
QMD – Quadratic Mean Diameter
R_h – Heterotropic Respiration
RCW – Red-cockaded Woodpecker
SDI – Stand Density Index
WSG – Western Gray Squirrel

Keywords

Carbon, Emissions, Fire, Forest Management, Mitigation, Sequestration

Acknowledgements

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Abstract

Objectives: The objective of this research was to use an ecosystem model to simulate the effects of management actions and disturbance on forest carbon dynamics. The central question was: What are the carbon tradeoffs of different management actions designed to meet different management objectives? Different management scenarios were evaluated at three different installations (Fort Benning, Camp Navajo, and Joint Base Lewis McChord), each with different management histories and a focal species of wildlife with specific habitat requirements.

Technical Approach: The project began with the intent of coupling an ecosystem model with a growth-and-yield model by using data on whole tree leaf biomass. However, for reasons including model performance and feasibility of technology transfer to installation managers, the project team transitioned to using a different modeling approach. The landscape-scale succession and disturbance model, LANDIS-II, was used because it performed well in validation and presents fewer challenges for technology transfer. The model uses an age-cohort based approach to simulate forest succession and growth over time and across space. In addition to the core model, the Century Succession extension was used, which enables the simulation of above and belowground carbon stocks and fluxes. The model framework was parameterized for each of the installations and used to simulated carbon dynamics as a function of different management scenarios.

Results: Increasing treatment intensity resulted in decreasing total ecosystem carbon. Treatments that included thinning, however, increased net ecosystem carbon balance, making thinned forests a larger sink for carbon. This finding is a result of decreased competition for resources (e.g. water, nutrients, light) resulting from thinning. At Fort Benning actively restoring longleaf pine forest decreased total ecosystem carbon by approximately 22% compared to the control (fire-suppressed broadleaved forest). Longleaf pine (*Pinus palustris*) restoration, however, increased red-cockaded woodpecker (*Picoides borealis*) habitat such that approximately 90% of the upland forest area (e.g. historically longleaf pine dominated forest) was viable habitat by the end of the simulation period. At Camp Navajo thinning and prescribed burning treatments decreased total ecosystem carbon relative to the control in the absence of wildfire. When wildfire was included in the simulations, however, the thin and burn treatment had higher total ecosystem carbon than the control by the end of the simulation period. This results from the reduced risk of high-severity wildfire from treatment. Simulations also demonstrated that thinning and burning decreases the risk of habitat loss for the Mexican spotted owl (*Strix occidentalis*). At Joint Base Lewis McChord treatments that included thinning resulted in the lowest total ecosystem carbon, but the highest net ecosystem carbon balance. In addition, thin-only and thin and burn treatments significantly increased the probability of Oregon white oak (*Quercus garryana*) presence by the end of the simulation period. This species provides an important food source for the western gray squirrel (*Sciurus griseus*) during mast years.

Benefits: Workshops were held at each of the installations to train resource managers in the operation of the parameterized models. The models provide a planning tool for evaluating the cumulative effects of stand-scale management actions on installation-wide forest carbon stocks and sequestration.

Objectives

Our objective was to develop a methodology for modeling forest carbon sinks and sources on Department of Defense (DoD) lands that can be used to quantify the carbon impacts of different forest management regimes, including those directed toward reducing fire emissions.

Technical Objectives

1. Develop a methodology for using a growth-and-yield model, integrated with a biogeochemical model, to characterize the carbon cycle for DoD forested ecosystems. This methodology will enable quantifying above- and below-ground carbon sinks and sources, their changes through succession, and their responses to management and disturbance, including prescribed burning.

2. Model and compare how changes in land-use practices, including ecologically-based forest management, plantation forestry, and biomass production, affect an installation's carbon footprint, as well as other ecosystem services (e.g.. biodiversity, wildlife habitat, water availability), over short (less than 10 years), intermediate (10 to 50 years), and long (greater than 50 years) time horizons.

3. Identify optimal forestry and land-use practices that balance maximal carbon storage with other ecosystem services (e.g. biodiversity and wildlife habitat), using a life-cycle carbon management assessment.

4. Quantify the carbon benefit that can be obtained using ecologically-based forest management to an installation's carbon footprint using the Climate Action Reserve forest protocol v3.1.

Technical Approach

Background

The forest carbon cycle can be generically described as a series of carbon pools that interact with each other and the atmosphere through a series of fluxes (Figure 1). Carbon is removed from the atmosphere through photosynthesis by trees and understory vegetation, and some of this carbon is released back to the atmosphere through respiration. Once sequestered, carbon in plant material eventually transitions to the snag, woody debris, or surface fuel pool for trees, and the surface fuel pool in the case of understory vegetation. During decomposition, some of the carbon from the dead plant material is incorporated into the organic horizon and mineral soil. In the absence of disturbance, the expectation is that the carbon stock grows to a theoretical maximum where Net Ecosystem Productivity (NEP, the annual change in the total ecosystem carbon stock) approaches zero (Hudiburg et al., 2009).

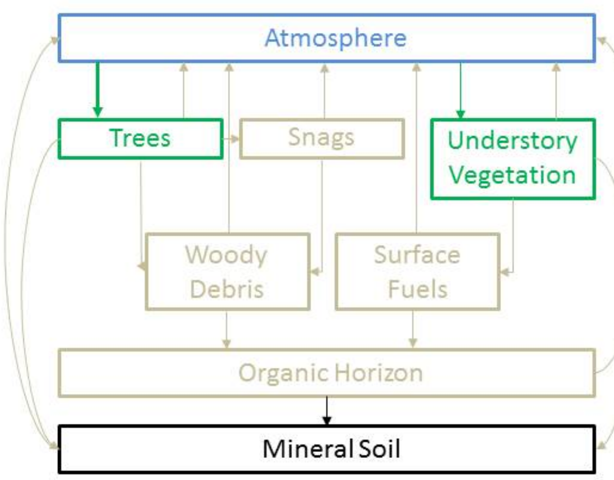

Figure 1: Generalized forest carbon pools and fluxes.

Figure 1: Generalized forest carbon pools and fluxes.

When fire enters into the equation, fluxes back to the atmosphere increase as a function of fire severity (Meigs et al., 2009; Wiedinmyer and Hurteau, 2010). Furthermore, increasing fire severity increases tree mortality (Agee and Skinner, 2005). When fire severity and tree mortality are high, NEP and carbon storage can decline precipitously (Dore et al., 2008; Meigs et al., 2009) and carbon stocks decline over longer time horizons as compared to forests experiencing lower fire severity (Hurteau and North, 2009; Hurteau, 2013). In fire-prone systems, the carbon carrying capacity - quantity of carbon that can be maintained under naturally prevailing conditions (Keith et al., 2009) - likely represents an appropriate target for carbon life-cycle management (Figure 2).

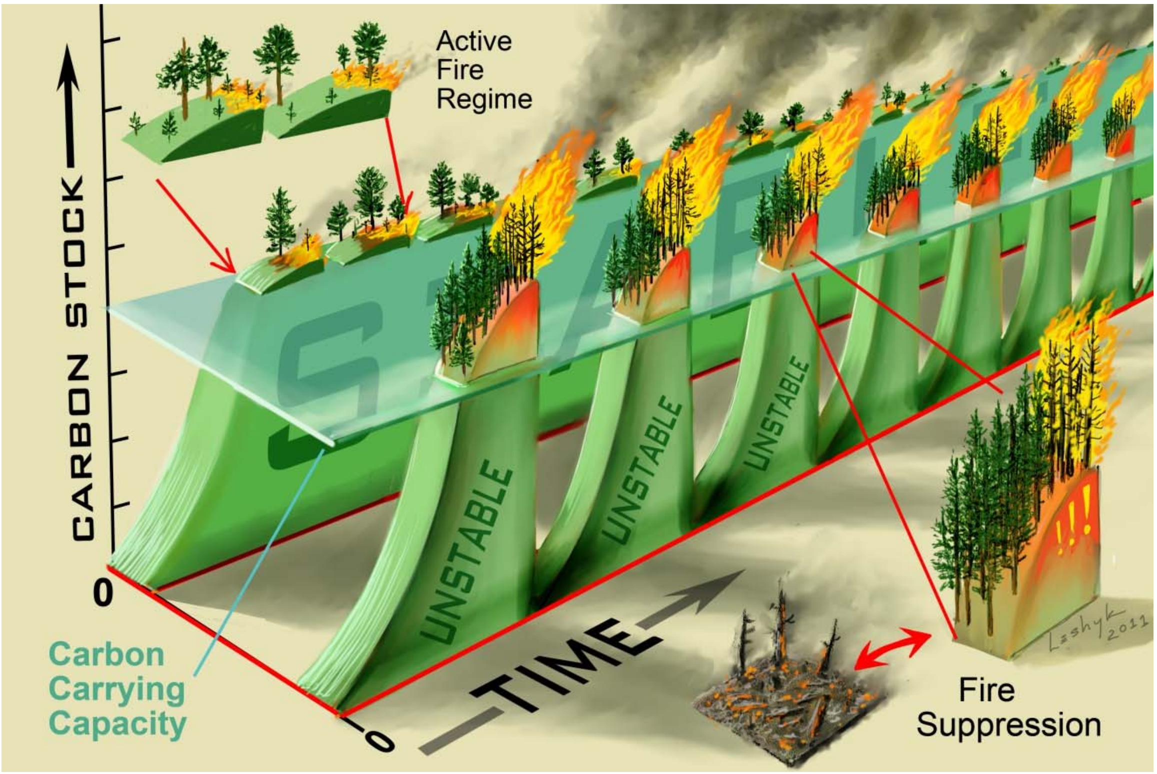

Figure 2: In dry, fire-prone forest types, the C stock varies as a function of the frequency of disturbance. An active fire regime results in a relatively stable C stock because frequent fires maintain fuel loads at levels that result in low-intensity fire. When wildfire occurs in the fire-excluded forest, the C stock is reduced below the carrying capacity. The C stock recovery following wildfire depends on the successional path of the forest recovery (from Hurteau, 2013).

Because forest carbon sequestration is a major part of the global carbon cycle (Canadell and Raupach, 2008), management actions that affect forest carbon dynamics are being increasingly scrutinized. In particular, the role of fire management, including its impacts on forest carbon and the resultant emissions is garnering substantial attention (Hurteau, 2013; Hurteau et al., 2013). Thus, quantifying the expected effects of management actions on forest carbon stocks prior to management implementation is increasingly important.

Forest management involves more than C stocks and fluxes, however, and over the last few decades traditional forest management and modeling has been modified to incorporate assessments of habitat conditions relevant to particular species. This has included efforts to identify and quantify stand structural characteristics such as coarse woody debris, multi-layered canopy structure, and shrubs that were not always included in inventory and assessment of forest biomass and timber stocks (Lindenmayer and Franklin, 2002). Now most forest inventories include assessments of these ecological attributes significantly improving models of listed or at-risk species habitat distribution and the influence of management actions on habitat quality. Recent advances in ecological forestry, however, have begun to recognize the importance of spatial structure across a range of scales in providing habitat characteristics that are not always assessed with metrics based on stand-level averages (Abella and Denton, 2009; Larson and Churchill, 2012). In particular, silvicultural efforts to maximize tree growth through regular spacing may produce tree patterns that lack an ecological analog (North et al., 2009; Larson and Churchill, 2012; Larson et al., 2012).

Analyses of forests that historically had a frequent fire regime have consistently shown that tree spatial patterns are highly irregular, creating a diversity of habitat and microclimate conditions. A recent western U.S. meta-analysis of all spatially-explicitly forest studies with reconstructions of historic tree patterns found a consistent within-stand pattern of three conditions; individual trees (I), clumps of trees (C) and openings (O) (Larson and Churchill, 2012). This ICO pattern creates high within-stand heterogeneity, a structure associated with preferred nesting and resting habitat for some listed or at-risk species, while also providing conditions for successful prey base foraging (North et al. 2010). Current efforts have focused on quantifying these patterns using new spatial analyses (2nd order point pattern analysis (Sanchez-Meador et al., 2011)) and translating relevant information into silvicultural prescriptions designed to create desired ICO patterns at appropriate scales (Churchill et al., 2013; Lydersen et al., 2013). The goal of our wildlife research is to identify these patterns at each of the three DoD installations, provide metrics for silvicultural prescriptions to create these conditions and assess how different scenarios for managing forest C stocks might affect these attributes.

Materials and Methods

To address the objectives of this research we used growth-and-yield and ecosystem models to quantify the effects of three different management objectives on forest C dynamics. The models were parameterized and validated with field data for three installations: Fort Benning, GA; Camp Navajo, AZ; Joint Base Lewis McChord, WA (Figure 3). The overall technical approach followed that outlined in Figure 4.

Fort Benning, Georgia is an approximately 75,000 ha military installation in the Sandhills ecological region (Dilustro et al., 2002). Soils are predominantly loamy sands or sandy loams (NRCS, 2013). National Climatic Data Center (NCDC) climate data collected for this study show an average maximum temperature of 33°C in July, minimum temperature of 2°C in January, and mean annual precipitation of 119 cm. The landscape includes forest types dominated by longleaf pine (*Pinus palustris*), mixed pines: longleaf, loblolly (*P. taeda*) and shortleaf (*P. echinata*), and mixed pine-hardwoods. The dominant species in the mixed pine-hardwoods type include southern red oak (*Quercus falcata*), blackjack oak (*Q. marilandica*), turkey oak (*Q. laevis*), and mockernut hickory (*Carya tomentosa*). The installation also includes plantations of longleaf and loblolly pine. Stands range in age from less than 10 years to greater than 100 years. Currently, 47% of stands are $\geq$ 60 years old, the minimum age to provide red-cockaded woodpecker (RCW) nesting habitat.

Fort Benning is designated as a core population site for the federally endangered RCW (Dilustro et al., 2002; USFWS, 2003). Habitat loss is the major cause of RCW decline. In addition to requiring pines at least 60 years old for nesting, they require low density forests with an open midstory for foraging (Lignon, 1970). RCWs prefer longleaf pine, although they will occasionally use older shortleaf and loblolly pines as cavity trees (Engstrom and Sanders, 1997). Frequent prescribed fire and regular thinning are integral components of the RCW habitat management plan (USFWS, 2003). Longleaf pine is fire-dependent and in the absence of fire every 2-10 years, it is outcompeted by more fire-sensitive hardwood species (Glitzenstein et al., 1995).

Camp Navajo is an approximately 11,500 ha National Guard facility located approximately 16 km west of Flagstaff, AZ. Elevation ranges from 2083-2364 m across the installation and soils are primarily derived from basalt, although sandstone and limestone parent materials occur at lower elevations. NCDC climate data collected for this study show an average maximum temperature of 26°C in July and minimum temperature of -12°C in December. The annual precipitation pattern is bi-modal, with a distinct dry period in May and June. Mean annual precipitation is 51 cm. The forest is dominated by ponderosa pine (*Pinus ponderosa*), interspersed with gambel oak (*Quercus gambellii*). The majority of stands are 60-70 years old, with some old-growth stands (>100 years).

Camp Navajo provides habitat for the federally threatened Mexican Spotted Owl (*Strix occidentalis*, MSO), which require structurally complex older forests for nesting and roosting. Logging from the late-19th through mid-20th century, coupled with fire suppression contributed to homogenization of forest structure and decline of the MSO. The stand-replacing fire risk of this current forest structure poses a major threat to the species. Management priorities for MSO focus on reducing the risk of stand-replacing wildfire by restoring the historical forest structure typical of southwestern ponderosa pine forest (USFWS, 2012).

Joint Base Lewis McChord (JBLM) is an approximately 36,183 ha installation located southwest of Tacoma, WA. Elevation ranges from 1-196 m across the installation. Approximately 50% of the soils on the installation are prairie soils in the Spanaway complex or a derivation thereof and the remaining portion of the installation is a mix of other soil types (NRCS Web Soil Survey). NCDC climate data collected for this study show a mean annual temperature of 12°C and mean annual precipitation of 143 cm. There are three predominant land cover classes at the installation; Douglas-fir forest (*Pseudotsuga menziesii*), mixed-forest with a substantial Douglas-fir component, and Willamette Valley upland prairie and savanna.

Joint Base Lewis McChord provides habitat for the western gray squirrel (*Sciurus griseus*, WGS), a species listed as threatened by the Washington Department of Fish and Wildlife since 1993 (WAC 232-12-011) and a federal species of concern. WGS habitat suitability is generally highest on the edge of prairies in the mixed forest with a Garry oak (*Quercus garryana*) and ponderosa pine component, habitat maintained by frequent fire. Efforts for bolstering the WGS population at JBLM include removal of Scotch broom (*Cytisus scoparius*) from prairies and thinning at the prairie-forest ecotone to promote oak and pine growth (Vander Haegen et al., 2007).

Figure 3: Longleaf pine forest (*Pinus palustris*, top), ponderosa pine forest (*Pinus ponderosa*, middle), Douglas-fir forest (*Pseudotsuga menziesii*, bottom).

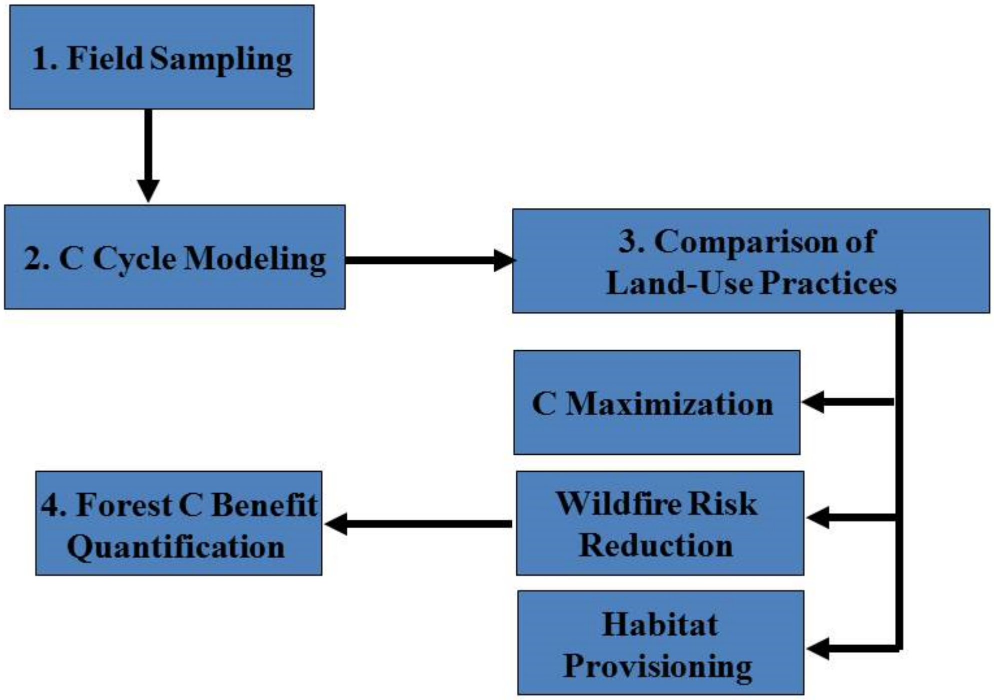

Figure 4: Work flow of the technical approach for project RC-2118.

Field Sampling:

Site selection for sampling was based on the objectives of this research, used input from natural resource managers at each installation, and was dependent on access as a function of training needs. Stratification utilized spatial data layers of forest type available at each installation. For each strata identified for sampling, all possible stands were selected and provided to training schedulers at each installation. Sampling from the population of stands for each stratum was determined by the scheduler at each installation. The number of plots sampled within each stratum was a function of the area occupied by the stratum within the boundary of the installation and access availability. Maps for each installation and the plot locations sampled are provided in Figures 5-7. The numbers of plots by stratum are listed in Table 1 for each installation.

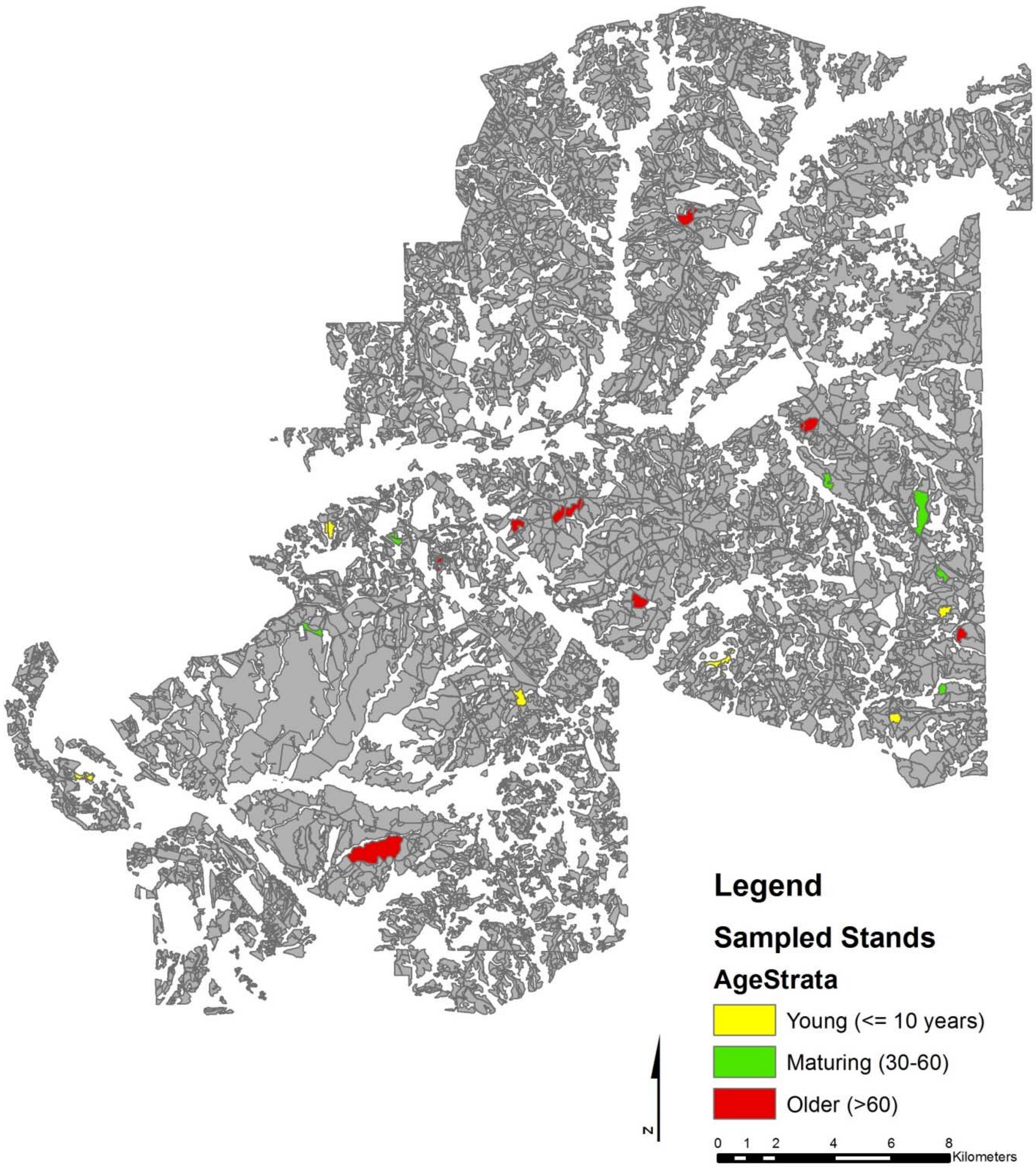

Figure 5: Map of Fort Benning with sampled stand locations by stratum.

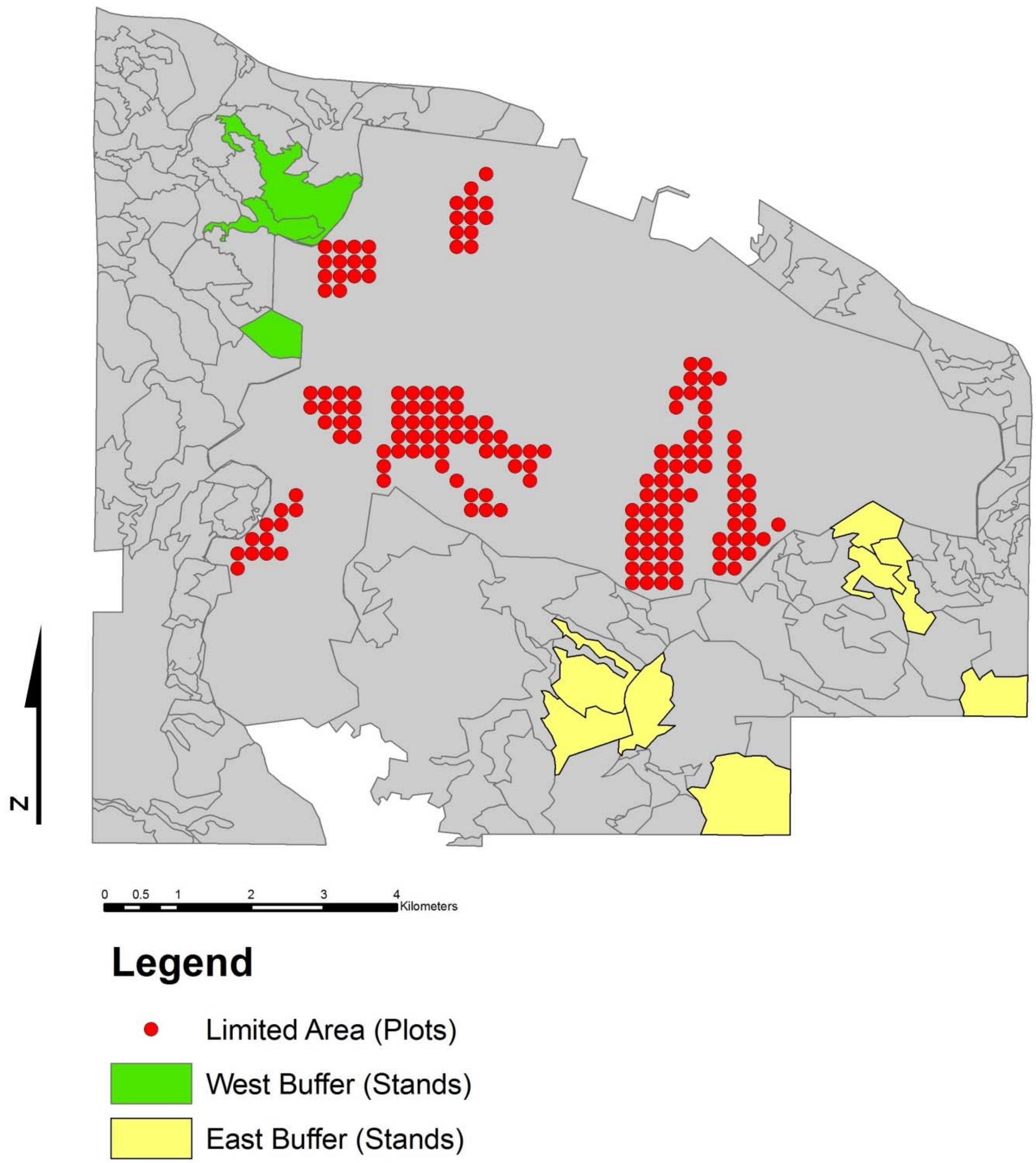

Figure 6: Map of Camp Navajo with sampled stand locations by stratum (East and West Buffer) and plot locations within the Limited Area. The East and West Buffer areas are part of the installation that provides a perimeter buffer for the secure Limited Area where the depot is maintained. The Limited Area does not have forest stands delineated because of the nature of the utilization of this area of the installation.

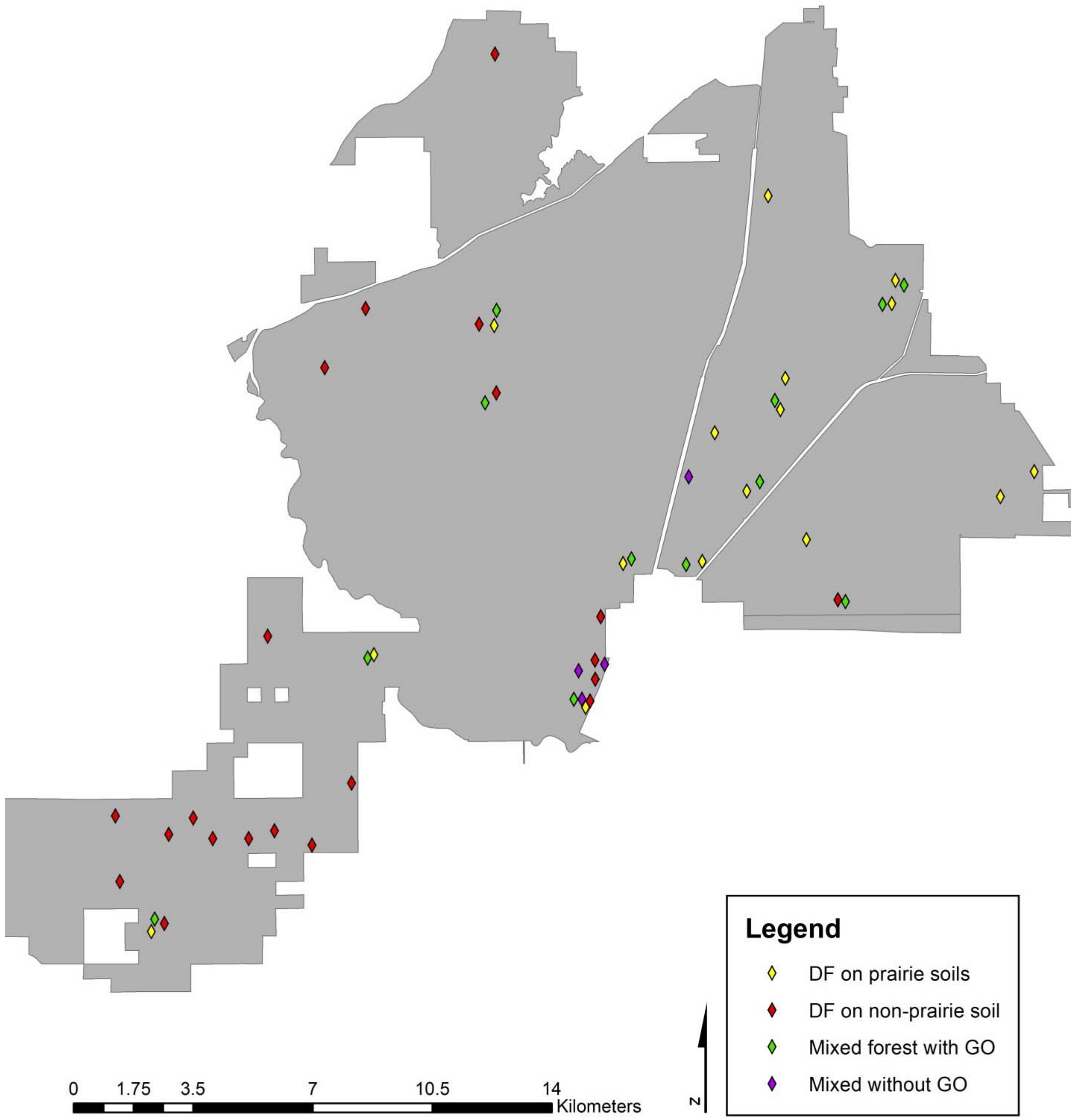

Figure 7: Map of Joint Base Lewis McChord with sampled stand locations by forest type.

Table 1: Numbers of plots by stand type at each installation.

Installation	Stratum	Number of Stands	Number of Plots
Fort Benning	Longleaf pine ≤ 10 years old	6	63
	Longleaf pine 30-59 years old	6	88
	Longleaf pine ≥ 60 years old	10	72
Camp Navajo	East Buffer, Pine	8	105
	West Buffer, Pine-Oak	3	51
	Limited Area, Pine	N/A	86
Joint Base Lewis McChord	FT1: ≥ 50% Douglas-fir, prairie soils	16	110
	FT2: ≥ 50% Douglas-fir, non-prairie soils	21	198
	FT3: mixed forest with Garry oak (*Quercus garryana*)	12	27
	FT4: mixed forest no Garry oak	4	11

Field sampling used the same nested plot design at all three installations (Figure 8). The only variation in sampling that occurred was the diameter cut-off for sub-plots at each installation. Diameter cut-offs were varied to account for variability in tree growth and were the same for Fort Benning (FB) and Camp Navajo (CN), but increased for Joint Base Lewis McChord (JBLM) (Table 2). The overall plot size was 1/5 ha (25.2 m radius) to adequately sample larger trees which occur with reduced frequency. Nested sub-plots (1/10 ha, 17.8 m radius; 1/50 ha, 8 m radius) were used to measure smaller diameter trees that occur with greater frequency in the forest. Tree specific measurements included species identification, diameter at breast height (DBH), total tree height, height to base of live crown, and health status. If a tree was dead, snag decay class was assigned following Maser et al (1979). Regeneration was tallied by 5 cm height class within a 2 m radius of plot center. At each plot three 15 m modified Brown's fuels transects were established to measure surface fuels and coarse woody debris (Brown, 1974). Along each transect one and ten hour fuels (0-0.64 cm and 0.64-2.54 cm) were tallied in the first 2 m, One hundred hour fuels (2.64-7.62 cm) were tallied along the first 4 m, and one thousand hour fuels (>7.62 cm) were measured along the entire transect. All 1000-hour fuels were classified on an I-V coarse woody debris decay class scale. Litter and duff depth measurements were made at 1 and 3 m along the transect. A subsample of plots was identified to collect data on whole-tree leaf biomass and soil carbon. This subsample was selected to capture the range of conditions sampled. At Fort Benning, leaf biomass samples were collected from 17 longleaf pine trees, representing the range of tree diameters present in the total sample of plots, and 12 soil cores were collected at two different depths in sand and sandy loam soil types. At Camp Navajo, leaf biomass samples were collected from 8 ponderosa pine trees, representing the range of tree diameters present in the total sample plots, and 12 soil cores were collected at two different depths from three different stands. At Joint Base Lewis McChord, leaf biomass samples were collected from 6 Douglas-fir trees, representing the range of diameters present in the total sample plots, and 12 soil cores were collected at two different depths from prairie and non-prairie soils.

Whole tree leaf biomass data were collected by harvesting branches, clipping all leaf biomass and weighing the leaf biomass using field balances. A subsample of leaf biomass was retained from each tree sampled, dried in the lab and re-weighed to determine dry weight of the leaf biomass. Regression equations were developed for the relationship between diameter at breast height and whole tree leaf biomass for each species. Soil samples were collected at 0-15cm and 15-30cm depths. Soil samples were processed at the Colorado Plateau Stable Isotope Lab to quantify total carbon.

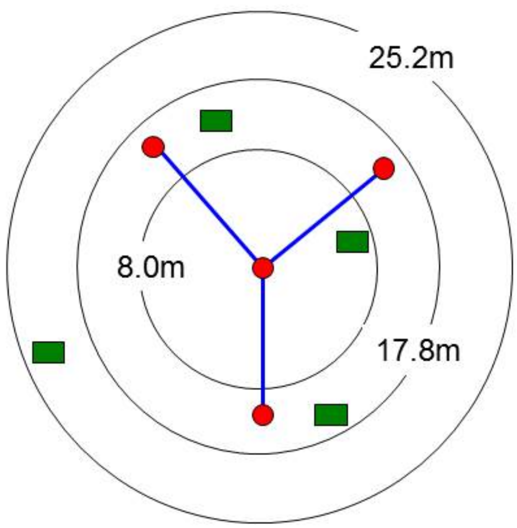

Figure 8: Field plot design used for collecting stand structure data. The plot structure included a 1/5 ha plot with 1/10 and 1/50 ha subplots. Specific diameter cut-offs for subplots are identified by installation in Table 2. Three modified Brown's fuel transects (blue lines) were installed from plot center. Soil samples (red dots) were collected from a subsample of plots. Whole tree leaf biomass was also collected from a subsample of trees within plots (green rectangles).

Table 2: Diameter at breast height (DBH) cut-offs used for sampling stand structure plots.

Installation	1/50 ha DBH	1/10 ha DBH (cm)	1/5 ha DBH (cm)
Fort Benning	≥5 cm	≥30 cm	≥50 cm
Camp Navajo	≥5 cm	≥30 cm	≥50 cm
JBLM	≥5 cm	≥50 cm	≥80 cm

Wildlife sampling methods:

Although methods and design varied between the three installations due to different listed or at-risk species and their preferred habitat conditions, the common focus was on analyzing within stand spatial patterns and scales and how trees might respond to treatment.

Red-cockaded woodpecker (*Picoides borealis*, RCW) habitat has been extensively studied to identify stand and landscape level attributes associated with preferred use (i.e., USFWS 1985, Zwicher and Walters 1999). The RCW is associated with mature, often large, diameter longleaf pines, and unlike other woodpeckers, excavates nests in live trees, often those with heart rot. Fort Benning has excellent inventories of the location (212 active clusters) and stand conditions associated with known nest locations. In general nesting conforms to the preferred habitat conditions identified; yet some forest stands with these attributes are not used. It does not appear that lack of use is due to birds failing to locate these stands, as woodpecker densities and dispersal of fledged young are both relatively high across the base's full extent. The goal of our research was to quantify within stand spatial structure in three different forest conditions: high quality habitat with woodpecker nests, habitat considered low quality yet with woodpecker nesting, and habitat considered high quality yet without nesting woodpeckers. We defined RCW habitat quality following the guidelines specified in the 2003 US Fish and Wildlife Service RCW recovery plan. Using the databases provided by The Nature Conservancy at Fort Benning, we identified potential sample stands for each of these three strata. In the field we inspected each stand, including it in the study if 1) nests could be located (usually tagged by base personnel) and 2) forest conditions were consistent with inventory assessment (i.e., species composition, stand age and tree size). For stands used by woodpeckers, we centered our plots on known nest cluster locations. For the high quality stands that were not used, we centered plots around large longleaf pine trees similar in size and age to nest cluster locations in the stands with woodpecker use.

At each plot, all trees and snags $\geq$ 5 cm DBH were identified, measured, and mapped (x and y coordinates) using a surveyor's total station. In addition the height to the base of the live crown and visible cavities were identified for each tree. Canopy cover was assessed using a densitometer (vertical sighting tube), and quantified as the percentage of 100 samples point where canopy foliage obscured the sky. Shrubs were measured within five 25 m^2 plots systematically located across the plot. All data were entered into Excel spreadsheets and then imported into ArcGIS for analysis. For each tree location, we modeled crown size based upon field measurements of each tree (height to crown and two crown radii). Individual trees, tree clumps and openings were identified using the methods described in Lydersen et al. (2013), a modification of the 2^{nd} order tree spatial statistics described by Larson and Churchill (2012).

At Joint Base Lewis McChord, a similar sample design and analysis were employed but with a focus on providing habitat for western gray squirrels (*Sciurus griseus*) (Gregory et al., 2010). Analysis of the western gray's movement and foraging patterns using radio telemetry have suggested an association with arboreal structure that provides interlocking or closely proximate tree crowns so that the squirrel does not have descend to the ground (Gilman 1986, Ryan and Carey 1995, Linders 2000, Gregory 2005). Observation and telemetry suggests crowns within a meter of each other may still be used because squirrels can jump that distance (Vander Haegen pers. comm.). At JBLM we used databases that identified known squirrel use areas and areas that had similar stand conditions but which squirrels had avoided (based on telemetry) or have never been recorded (observations). We also required the avoided stands to have squirrels in adjacent forested areas to avoid sampling habitat that might be suitable but that squirrels have not reached. Stands avoided by squirrels were areas that had been thinned. We, therefore, recorded stump locations, sizes and species so as to reconstruct forest stand conditions that existed before stand treatments (see North et al. 2007 for methods). In each stand we mapped all trees and snags $\geq$ 5 cm DBH within a 50 by 50 meter plot. Canopy cover was recorded with a densitometer. The crown projection of each measured tree was modeled using field measurements of base to the height of the live crown and measurements of two perpendicular crown radii.

At Camp Navajo, ponderosa pine forests are treated for fuels reduction by thinnings designed to produce ICO patterns that may develop into habitat for the Mexican spotted owl (*Strix occidentalis*). Although most of the CN forests are presently too young and trees are too small to provide preferred habitat conditions, treatments are designed to accelerate tree growth while producing the clumps, individual trees and openings found in older ponderosa pine forests that are used by owls. Here the question was the tradeoff between thinning to accelerate the development of large trees versus the possible reduction in growth rate that may occur from leaving trees in clumps rather than evenly spaced. Installation personnel stated that a high research priority was improving silvicultural and fuels reduction treatments by identifying optimal tree clump and gap size that facilitated rapid tree growth while still providing within stand spatial heterogeneity associated with owl use.

At Camp Navajo, we identified fuels treated areas where mechanical thinning had reduced stand densities and left an ICO pattern. Within these areas we selected tree clumps and identified, measured, and mapped all trees within a 6 m radius around a 'target' tree. Each target tree was measured and then two increment cores were taken at breast height and 90^0 to each other. Cores were mounted, sanded, and crossdated using standard methods (Speer, 2010). Annual radial increment growth was measured using a scanner and WinDendro software (Regents Instruments). Each tree's annual basal area increment (BAI) was calculated averaging the two cores. We compared average BAI for the five years before the thinning with the 5 year average BAI starting two years following thinning treatments to allow for growth response to stabilize. To adjust for potential growth differences due to climate rather than change in stand conditions, we used a correction factor calculated as the average Palmer Drought Severity Index (PDSI) values for the post-treatment 5 year period (starting 2 years after treatment) divided by the pre-treatment 5 year period taken from the nearest gridpoint (88; Lat. 35°N, Long. 112.5°W). Stand density around each target tree was calculated using the mapped locations of all the trees within the 6 m radius and the Hegyi index (Das et al., 2008). This is a weighted index that reflects the relative crowding around a target tree accounting for the distance and size of trees surrounding the target tree. Das et al. (2008) showed it is a more effective measure of localized, relative density than Theissen polygon analysis. We analyzed how radial increment growth changed following thinning treatments across a range of clumps sizes and densities to identify possible size and density thresholds that reduced growth response.

Analysis of data related to the wildlife habitat portion of this research involved the calculation of two different metrics of forest structure. Quadratic Mean Diameter (QMD) is a widely used measure in silviculture that assesses the central tendency of tree diameters rather than the arithmetic mean. The difference is that QMD accounts for basal area which has been shown to be a better measure of ecological resource use. QMD is calculated as:

$$QMD = \sqrt{\frac{BA}{k * n}}$$

where BA is the stand basal area, n is the number of trees, and k is a constant based on measurement units (k = 0.00007854 for BA in m^2). The number of continuous crown paths is an attribute used to quantify connectivity. To quantify the number of continuous crown paths we used an algorthim in ArcGIS that connects tree polygons and then counts the number of uninterrupted lines (paths) that reach across the plot.

Modeling:

Growth-and-yield models are useful for predicting forest response to different silvicultural manipulations and the effects on aboveground carbon. However, aboveground C only represents a portion of total ecosystem C. Quantifying the effects of management actions on total ecosystem C requires the use of a biogeochemical model to capture belowground C dynamics. We had originally proposed to use the Forest Vegetation Simulator in tandem with Biome-BGC to simulate the effects of treatment on ecosystem carbon dynamics by using the relationship between DBH and leaf biomass to use the forest vegetation simulator (FVS) outputs to inform leaf biomass parameter values in Biome-BGC to capture the effects of thinning. However, given Biome-BGC's lack of a developed user-interface, we foresaw some difficulty in the tech transfer phase of this project. Consequently, we used LANDIS-II and the Century Succession extension as a platform for evaluating the effects of silvicultural manipulations on total ecosystem carbon. Point-level comparisons of the FVS/Biome-BGC approach and LANDIS-II approach were run for longleaf pine stands at Fort Benning. We found that the FVS/Biome-BGC approach under-predicated total ecosystem carbon and that the LANDIS-II approach showed no directional bias (RMSE 13.6 Mg C ha^{-1}).

FVS is a growth and yield model that can simulate a wide range of silvicultural treatments for most major forest tree species, forest types, and stand conditions (Crookston and Dixon, 2005). Since its initial development in 1973, the basic FVS model structure has been calibrated based on decades of natural resources research. Because of its applicability to a wide range of treatments and forest stand conditions, FVS is the most widely used growth-and-yield model in the United States. The Fire and Fuels Extension (FFE) of FVS expands on FVS by incorporating models of fire behavior and effects, snag dynamics, and fuel accumulation. FVS simulates understory plant growth and its contribution to fuels as a function of forest type and canopy cover to determine dry weight per unit area (Rebain et al., 2010). A new sub-model has been added that tracks carbon pools (Rebain et al., 2010). The carbon submodel uses genus-specific allometric equations from Jenkins et al. (2004) to calculate live tree carbon. This added flexibility allows us to track the effects of prescribed fire on life-cycle carbon management. FVS-FFE also provides the flexibility to simulate both wildfire and prescribed fire at user specified points or random points in time and track potential fire behavior and effects without initiating a fire event (Reinhardt and Crookston, 2003).

Biome-BGC is a process-based model that uses meteorological data, site characteristics, and vegetation parameter values to quantify fluxes of carbon, water, and nitrogen (White et al., 2000). The current version of Biome-BGC allows for the quantification of fire impacts on system fluxes and has been used to compare the effects of disturbance history on a number of coniferous forest types (Thornton et al., 2002). FVS outputs of forest structural attributes will be used to inform changes in leaf biomass in Biome-BGC to simulate carbon fluxes using this process-based model. We will estimate the amount of stand-level leaf biomass as a function of basal area and diameter distribution from FVS using the field data-derived linear relationships between diameter at breast height and species for each of the installations. Changes in leaf biomass as a function of changes in forest structure will be altered in Biome-BGC to effect change in the leaf area of the simulated system.

The LANDIS-II model is a spatially explicit forest simulation framework for simulating forest succession and disturbance across large landscapes (Scheller and Mladenoff, 2004). The landscape is divided into a series of interacting cells that can vary in size from meters to kilometers. The model uses an age cohort approach to modeling forest tree species distribution and growth. During succession, age-related mortality, reproduction, and cohort aging are simulated. In the event of a disturbance a new cohort is established, with mature cohorts in neighboring cells serving as the seed source. The framework will accommodate a site having multiple cohorts of a single species. LANDIS-II allows for the incorporation of ecosystem processes operating at different time-steps in the simulation framework (Scheller et al., 2007), inclusive of a biomass module that is linked to the process-based model PnET-II (Scheller and Mladenoff, 2004). In addition to the core model function, LANDIS-II has a biogeochemical model extension called Century Succession, which incorporates the Century model into the LANDIS-II framework to simulate belowground carbon dynamics (Scheller et al., 2011a). Within the model a landscape simulation area is divided into ecoregions. Ecoregions are differentiated by soil and climate properties (Ravenscroft et al., 2010). The ecoregion module also includes species-specific probability values for establishment (Scheller et al. 2007), an important attribute for this proposed research. Each ecoregion is divided into a series of interacting cells in which cohort dynamics are simulated. A species input file is required to define species-specific life history attributes that influence cohort regeneration, development, and mortality.

In the LANDIS-II Dynamic Fire and Dynamic Biomass Fuels extensions used for simulation fire in this project, fire behavior and effects are estimated using fuels and climate data using methodology adapted from the Canadian Forest Fire Behavior Prediction System (Van Wagner et al., 1992; Sturtevant et al., 2009). We adjusted the fuel parameters from conifer fuel types in the Canadian Forest Fire Behavior Prediction System using fuel models from Scott and Burgan (2005), field data collected as part of this project, and values from the fire ecology literature for each installation. Parameters were tested and considered to be representative of wildfire effects based on the resulting fire severity and mortality across the simulated landscape over the course of the simulation period for each installation. We used the LANDIS-II Leaf Biomass Harvest extension to simulate thinning treatments. The effect of thinning treatments is accounted for in the Century Succession Extension as a function of the amount of wood and leaf biomass removed during harvest. The proportion of each species-age cohort harvested that remains is the system is allocated to different dead carbon pools (e.g. leaf, wood, roots) and the effect of its decomposition is accounted for in NECB.

Model Comparison of Land-use Practices:

We simulated three potential land management practices at each installation: 1) carbon maximization, 2) wildfire risk mitigation, and 3) threatened/endangered species habitat provisioning. To develop a range of potential management actions that are ecologically appropriate for each forest type for simulating these three scenarios, we used a combination of literature reviews and discussions with managers at each installation. The installation specific treatments varied as a function of the scenario objective.

Fort Benning FVS Treatments

At Fort Benning, we simulated control, prescribed burning, thinning, and combined thin and burn treatments. The default fuel values in FVS were reparameterized with field data and the bulk density values of longleaf pine litter and duff determined by (Parresol, 2005). In the control simulation, no management actions were implemented but regeneration of all species recorded in the field regeneration plots was added every five years for the first 20 years of the simulation to reflect seedling dynamics until canopy closure. Each time regeneration was added, the number of seedlings per species was decreased by 25% due to the decreasing light availability.

Prescribed burning was implemented on a three year rotation to reflect current management at FB and throughout the region to promote continued dominance of open canopy longleaf pine. Burns were assigned under simulated weather and fuel conditions that reflect prescribed fire during the spring, including windspeeds of 12.8 kph at 6 m above the vegetation, temperatures of 21°C, and fuels were either coded as wet or dry. Dry fuels were used in some cases to ensure fire ignition in all stands, and fires were simulated to spread to 70% of the stand area. The same prescribed fire parameters were also used at Camp Navajo because these values fall within acceptable prescription ranges in both forest types. Fire-induced mortality estimates were found to be ecologically inaccurate for prescribed fire, particularly in the case of longleaf pine. Therefore, the automatic mortality calculation was turned off and replaced with a simulated mortality for small trees that maintained an ecologically realistic number of trees throughout the 100-year simulation period (2% mortality for all trees 0-10.16 cm DBH in each fire). Regeneration of all species following each fire was based on field regeneration measurements in young (≤ 10 yrs), maturing (30-59), and older (≥60 year) stands.

Thinning prescriptions were implemented in year 2041 in the thin-only and in 2061 in the thin and burn, when the majority of young (initially ≤ 10 yrs) approached a stand density index (SDI) of 250, a point at which thinning is recommended to create future RCW habitat (Shaw and Long, 2007). The thinning prescription was implemented to reduce BA to 18 m^2 ha^{-1} and was implemented in all stands that exceeded this density, including stands > 10 years old initially. This BA target was selected because the southern variant of FVS does not model crown fire. This BA target has been found to modify fire behavior in southwestern ponderosa pine forests and was selected because of the structural similarities between fire-maintained longleaf pine and ponderosa pine forests. Regeneration was added following thinning using the same number of seedlings as a prescribed burn. The thin and burn treatment used the same conditions for prescribed fire and thinning combined into one simulation.

Camp Navajo FVS Treatments

Treatments simulated at CN included control, prescribed burning, thinning and combined thinning and burning. FVS default fuel values were updated with field measured values. Prescribed burning was implemented every ten years, which is within the historic mean fire return interval for southwestern ponderosa pine in northern Arizona (Covington and Moore, 1994; Covington et al., 1997). Prescribed fires were implemented in FVS using weather conditions typical of prescribed fire application in the region: windspeeds of 12.8 kph at 6 m above the vegetation, temperatures of 21°C, dry fuels, with fires burning 70% of the stand.

Thinning treatments were implemented at the beginning of the simulation due to the density of stands at CN. The thinning prescription used a thin-from-below to BA target of 18 m^2 ha^{-1}, which is commonly used to reduce fire risk and push the forest toward conditions more typical of the historical fire-maintained forest condition (Fulé et al., 1997; Fulé et al., 2001). The thin and burn treatments used a combined initial thin, followed by prescribed burning every ten years. After each treatment that increased seedling resource availability (thinning or burning), regeneration of ponderosa pine and gambel oak (*Quercus gambelii*) was added based on field measurements, literature, and the scale of disturbance (ie. greater regeneration in the thin and burn when compared to burn only).

Joint Base Lewis McChord FVS Treatments

Treatments simulated at JBLM included control, prescribed burning, thinning, and thinning coupled with burning. Field measured fuel values replaced default FVS values. Regeneration was prescribed at each time-step based on field measured values. Prescribed burning was simulated every 15 years using windspeeds of 12.8 kph at 6 m above the vegetation, temperatures of 21°C, dry fuels, with fires burning 70% of the stand. Because prescribed burning is relatively rare in this area (primarily in prairies), we applied the same prescription as CN. The thinning treatment at JBLM was designed to mimic current harvesting practices where 15% of the basal area is harvested every ten years. The thin and burn treatment included harvesting at 10-year time intervals with prescribed burning implemented every 15 years.

We also simulated targeted restoration of oak savannah on prairie soil types. The purpose of these simulations was to examine the C consequences of restoring habitat for the western gray squirrel. We identified a 708 ha parcel that had prairie soil and extant oaks. The simulation included complete harvest of Douglas-fir, thinning ponderosa pine from below, and prescribed burning. The objective of the thinning treatments was to increase light availability for oak regeneration and reduce competition for mature oaks.

Results and Discussion

Field data summary

Diameter distributions varied by stratum within each installation (Figures 9-11). Current conditions were also influenced by past management actions. Carbon stocks varied by stratum at each installation (Table 3) and were calculated using genus specific allometric equations from Jenkins et al. (2004).

Table 3: Live tree carbon by stratum for each installation. Standard error is presented in parentheses.

Installation	Stratum	Live C (Mg ha^{-1})
Fort Benning	$\leq$ 10 years old	16.2 (2.7)
	30-59 years old	29.6 (5.5)
	$\geq$ 60 years old	33.3 (2.4)
Camp Navajo	East Buffer, Pine	70.7 (6.0)
	West Buffer, Pine-Oak	50.1 (2.4)
	Limited Area, dense Pine	58.1 (5.1)
Joint Base Lewis McChord	FT1: $\geq$ 50% Douglas-fir, prairie soils	189.8 (7.2)
	FT2: $\geq$ 50% Douglas-fir, non-prairie soils	196.2 (4.8)
	FT3: mixed forest with Garry oak	151.1 (9.8)
	FT4: mixed forest no Garry oak	65.2 (12.1)

The relationship between whole tree leaf biomass and DBH was developed for the dominant species at each installation (FB = longleaf pine, CN = ponderosa pine, JBLM = Douglas-fir). Results show that DBH captures the majority of the variation in whole tree leaf biomass (Figure 12).

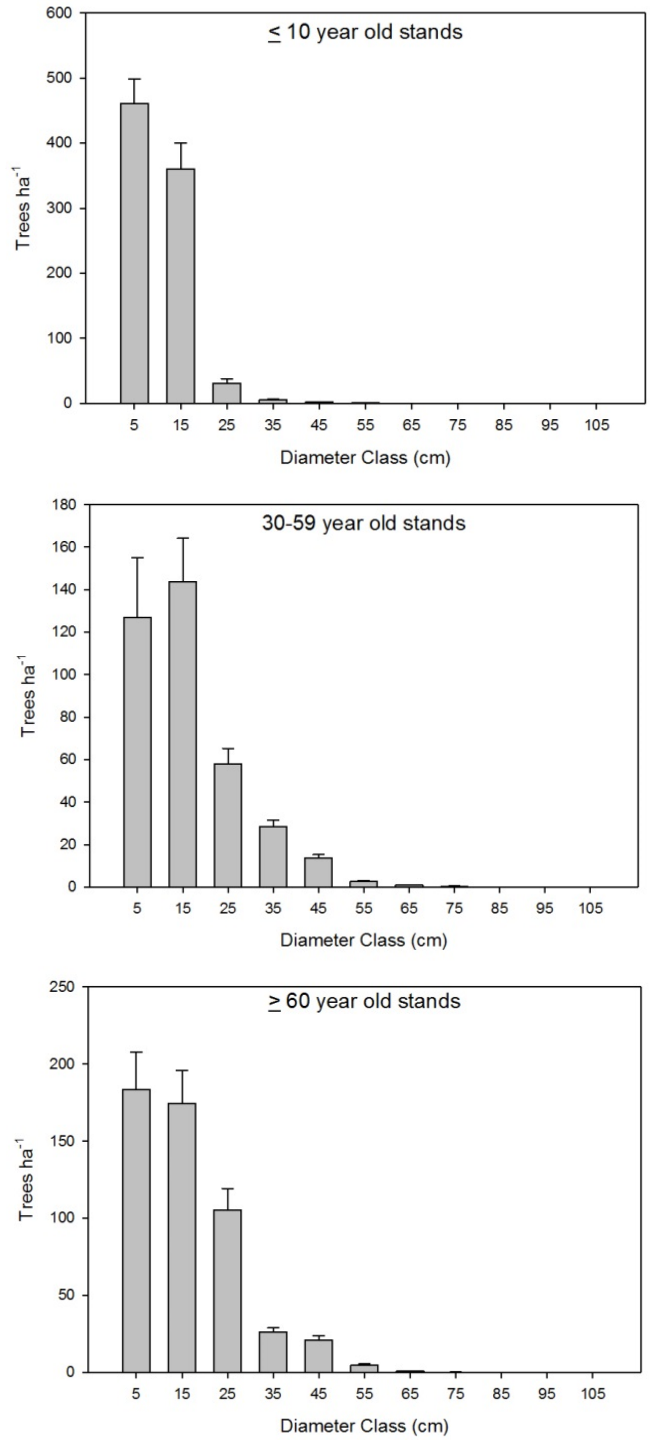

Figure 9: Diameter distributions for each stratum at Fort Benning. Note that the y-axis varies by stand type.

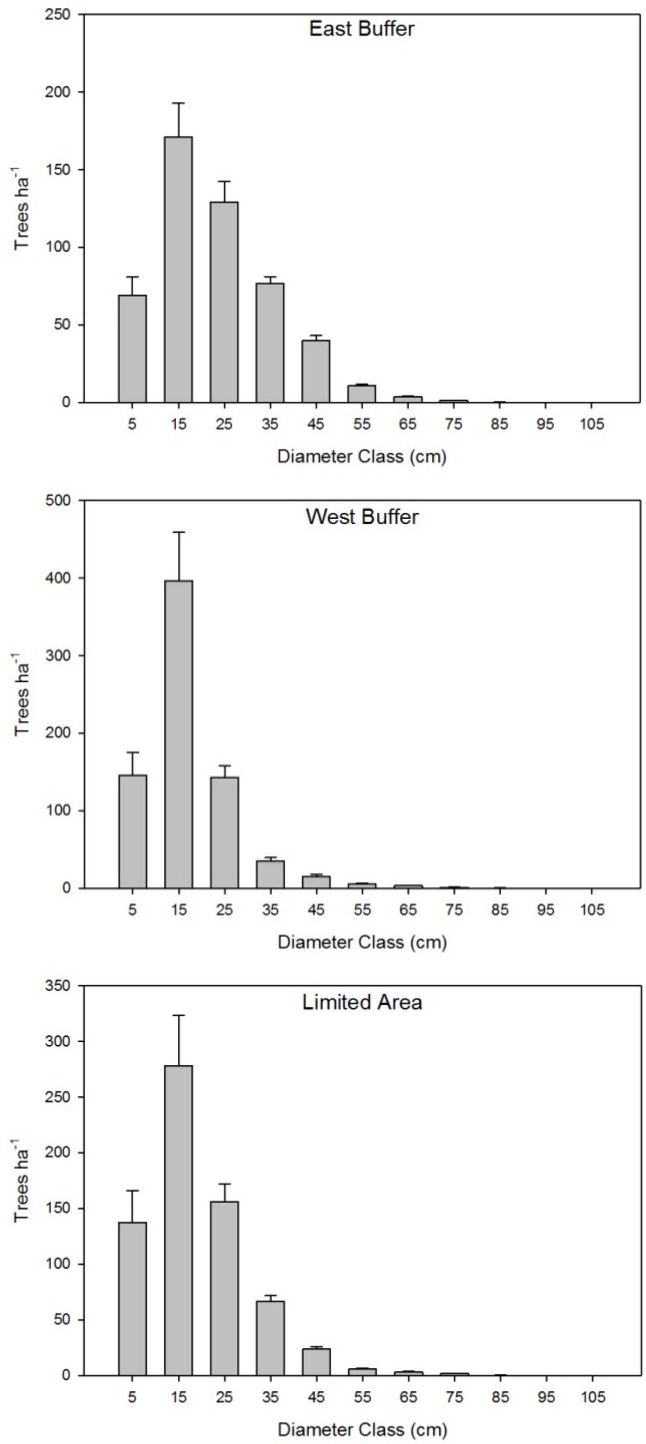

Figure 10: Diameter distributions for each stratum at Camp Navajo. Note that the y-axis varies by stand type.

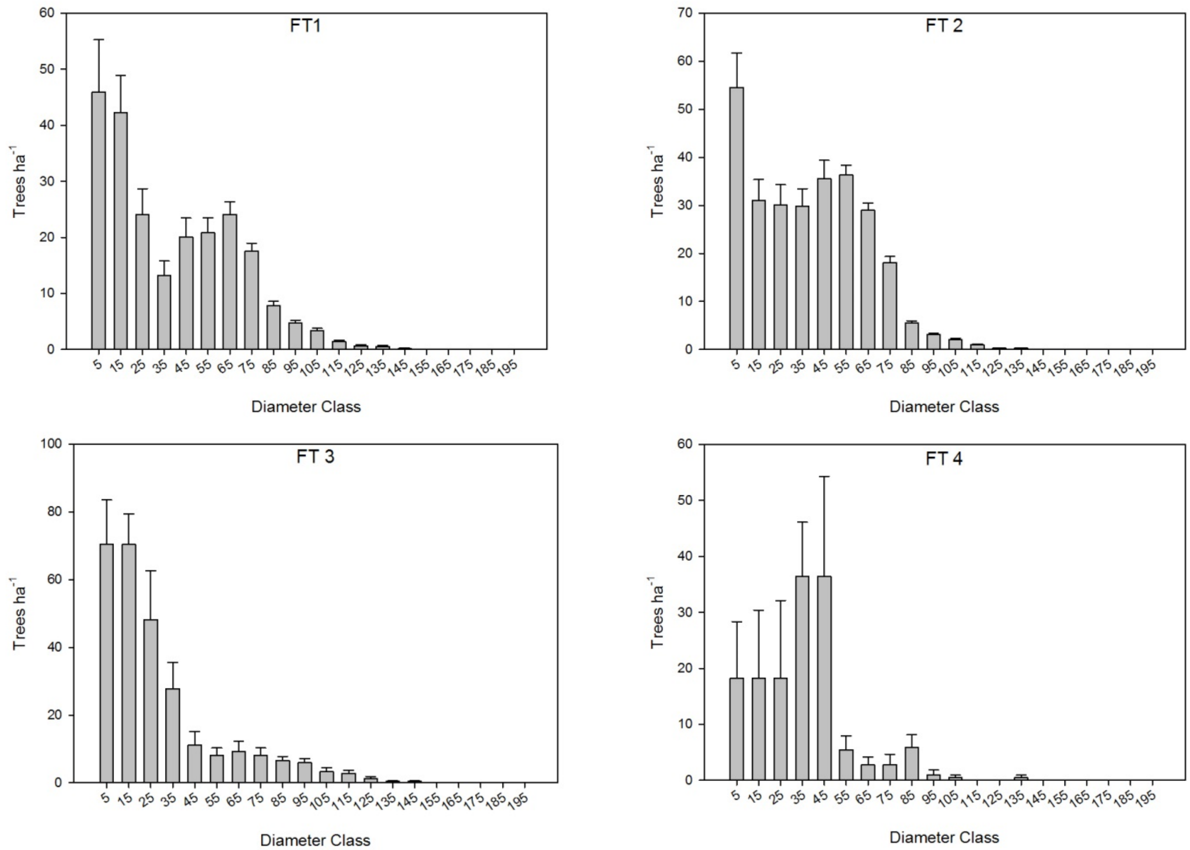

Figure 11: Diameter distributions for each forest type (FT) at Joint Base Lewis McChord. FT1: > 50% Douglas-fir on prairie soils; FT2: > 50% Douglas-fir on non-prairie soils; FT3: mixed forest with Garry oak; FT4: mixed forest without Garry oak. Note that the y-axis varies by forest type.

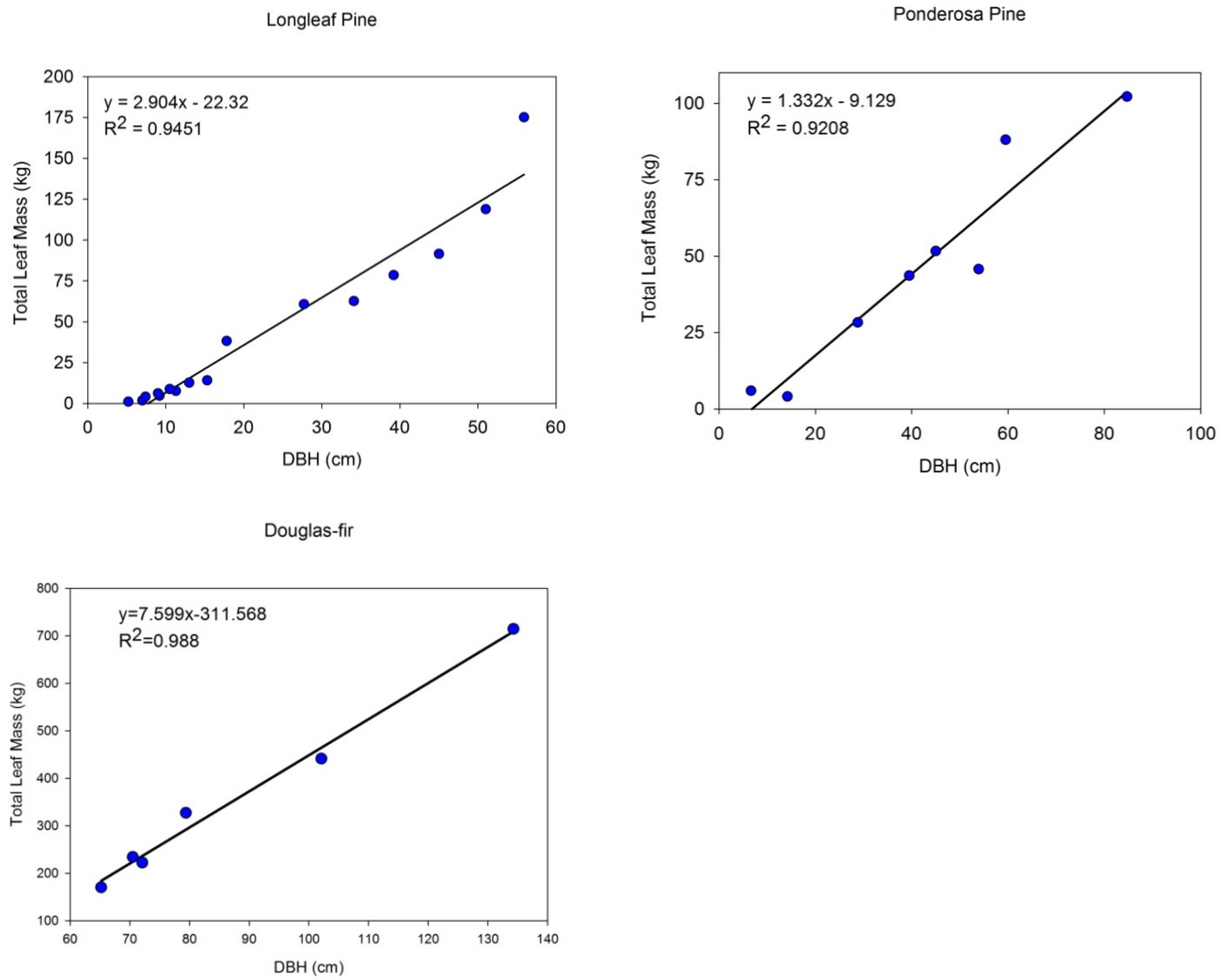

Figure 12: The relationship between whole tree leaf biomass and diameter at breast height for the dominant species at Fort Benning (longleaf pine), Camp Navajo (ponderosa pine), and Joint Base Lewis McChord (Douglas-fir).

FVS Simulations

Simulations were run using the Forest Vegetation Simulator for each of the installations by stand type. Simulations were implemented using field data for all treatments and run over a 100-year period. Results for live tree, snag, and total stand carbon for FB, CN, and JBLM are presented in Figures 13, 17, 21. Diameter distributions at simulation year 50 are presented in Figures 14-16, 18-20, 22-24.

Fort Benning FVS Simulations

At FB, the influence of treatments on C dynamics varied as a function of stand age. By the end of the 100-year simulation, live tree C in the young stands with simulated prescribed burning, thinning, and thin/burn were not significantly different. End-of-simulation C stocks in the control were approximately 25 Mg C ha^{-1} larger than the other treatments. The 30-59 year old stands had a divergence in live tree C stocks over the simulation period. The control had the largest increase, followed by the thin-only treatment. The burn-only and thin and burn treatments were not significantly different from each other and were lower than the thin-only over the majority of the simulation period. The same pattern was present for stands $\geq$ 60 years old. However, the difference between the thin-only treatment and the treatments that included burning was much less. The same general patterns persist when examining the total stand C (Figure 13). The impacts of simulated treatments on forest structure varied by stratum, with young stands ($\leq$ 10 years old) responding differently than mid (30-59 years old) and old ($\geq$ 60 years old) stands. In young stands, the control, burn-only, and thin/burn treatments achieved stand structures similar to diameter distributions for field measured old stands (Figure 14). Mid and old stands responded well to treatments that included burning, with regard to stand characteristics that are conducive to providing Red-cockaded woodpecker habitat. By the middle of the simulation period, stands in these two strata were lower density and comprised of larger diameter trees (Figures 15-16).

Camp Navajo FVS Simulations

Treatment influences were consistent across strata at CN due to the relatively homogenous nature of the forests at this installation (Figure 17). The control treatment consistently had the highest live tree and total C stocks across the simulation period. While the thin-only treatment resulted in a substantial reduction in C when implemented, end-of-simulation C stocks approached those of the control simulations. The burn-only and thin and burn treatments were consistently lower by approximately 50-60 Mg C ha^{-1} than the control and thin-only treatments by the end of the simulation period (Figure 17). Treatments at CN had similar effects on stand structure across strata (Figures 18-20). Thinning early in the simulation period reduced the number of trees per hectare, but regeneration resulted in the thin-only treatment diameter distribution exhibiting structural characteristics similar to the control by year 50 of the simulation. Treatments that included burning pushed stands toward dominance by larger diameter trees by simulation year 50. Large numbers of trees in the 5 cm diameter class in the burn-only treatments are a function of regeneration and prescribed fire initiation timing within the simulation. Subsequent prescribed fires in the burn-only treatment reduce the number of individuals in the smallest diameter class.

Joint Base Lewis McChord FVS Simulations

Forest types 1-3 had similar responses to treatments (Figure 21). The control simulations maintained the largest live and total C stocks throughout the simulation period. Live tree C stocks in the burn-only treatment were nearly as large as live tree C stocks in the control. Thin-only carbon stocks showed a slight increase throughout the simulation period, which was expected since the treatment was designed to provide a sustainable yield. Thinning coupled with regular prescribed burning resulted in slight declines in carbon stocks over the simulation period due to the effects of regular burning on regeneration.

Forest type 4 had fairly consistent C stock reductions with increasing treatment intensity. Overall, C stocks are approximately one-third lower in FT 4 than in the other forest types. One primary difference between forest type 4 and the other forest types was the response to the thin-only treatment. Regular thinning maintained fairly consistent C stocks over the simulation period in the other forest types. However, in FT4 the thin-only treatment increased by approximately 50 Mg C ha^{-1} over the course of the simulation period. Diameter class distributions showed a similar response in stand structure to treatments across strata (Figures 22-25). The control and thin-only treatments show an inverse-J distribution typical of forests in the region. Simulations that included regular burning resulted in a distribution with a larger number of trees in the mid-sized diameter classes by year 50 in the simulations. Simulations that included burning also had a much smaller number of trees per hectare because of the control that regular prescribed burning exerts on regeneration.

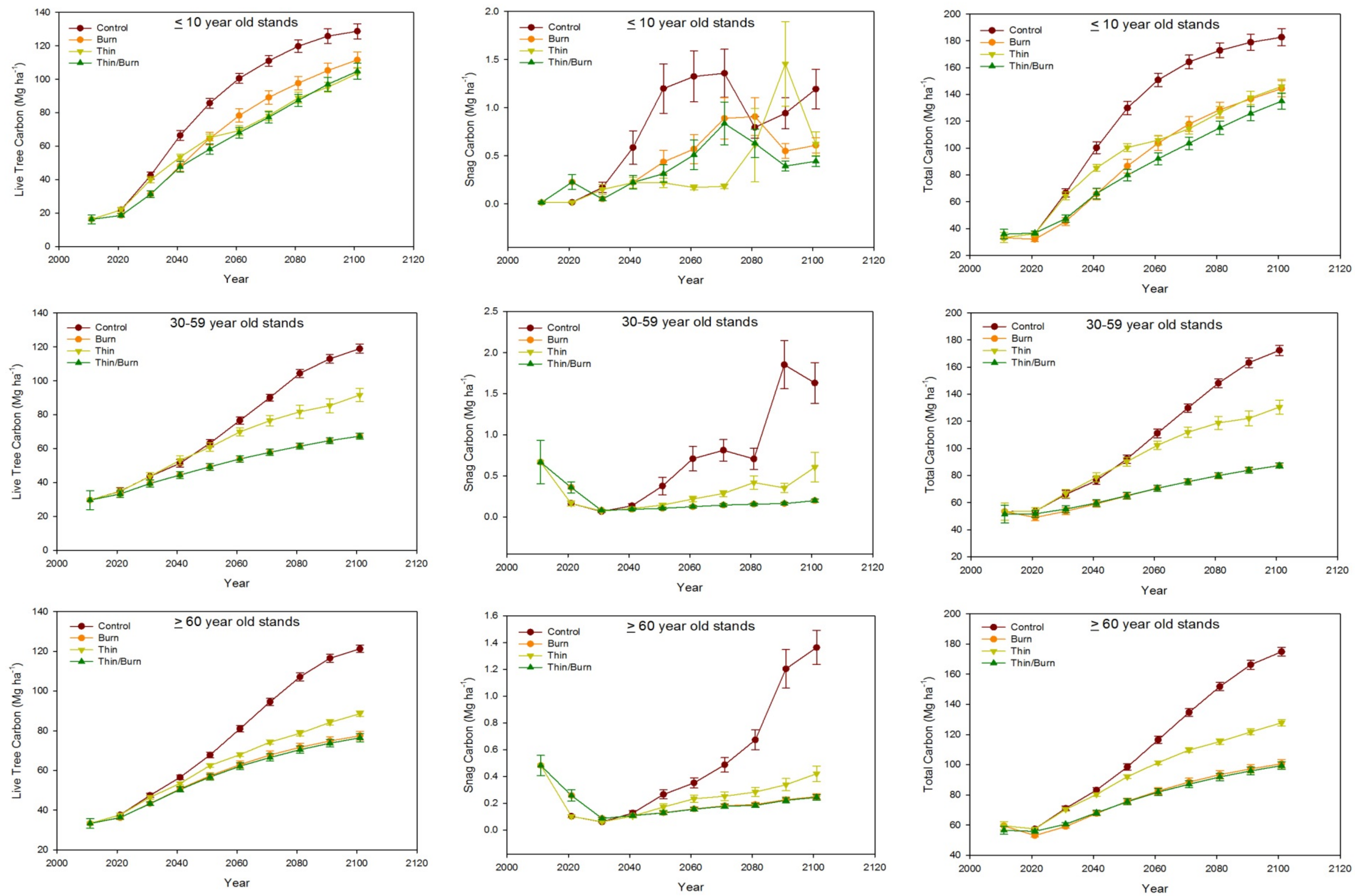

Figure 13: Live tree carbon (first column), carbon in snags (second column), and total stand carbon (third column) at Fort Benning. The first row is young stands, the second row is stands 30-59 years old, and the third row is stands > 60 years old. Note the y-axis scale varies among graphs.

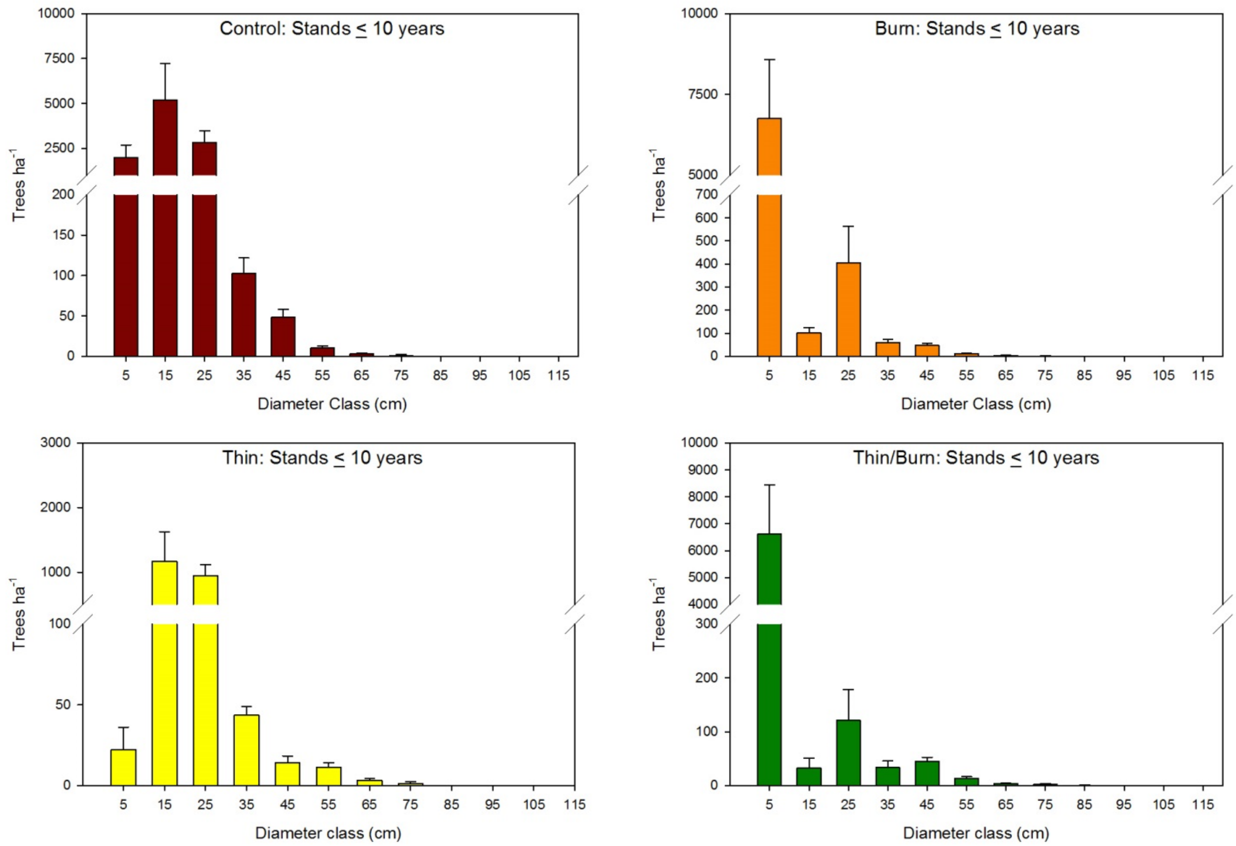

Figure 14: Year 50 simulated diameter distributions for stands with initial conditions 10 years old and younger at Fort Benning by treatment. Note that the y-axis varies by treatment.

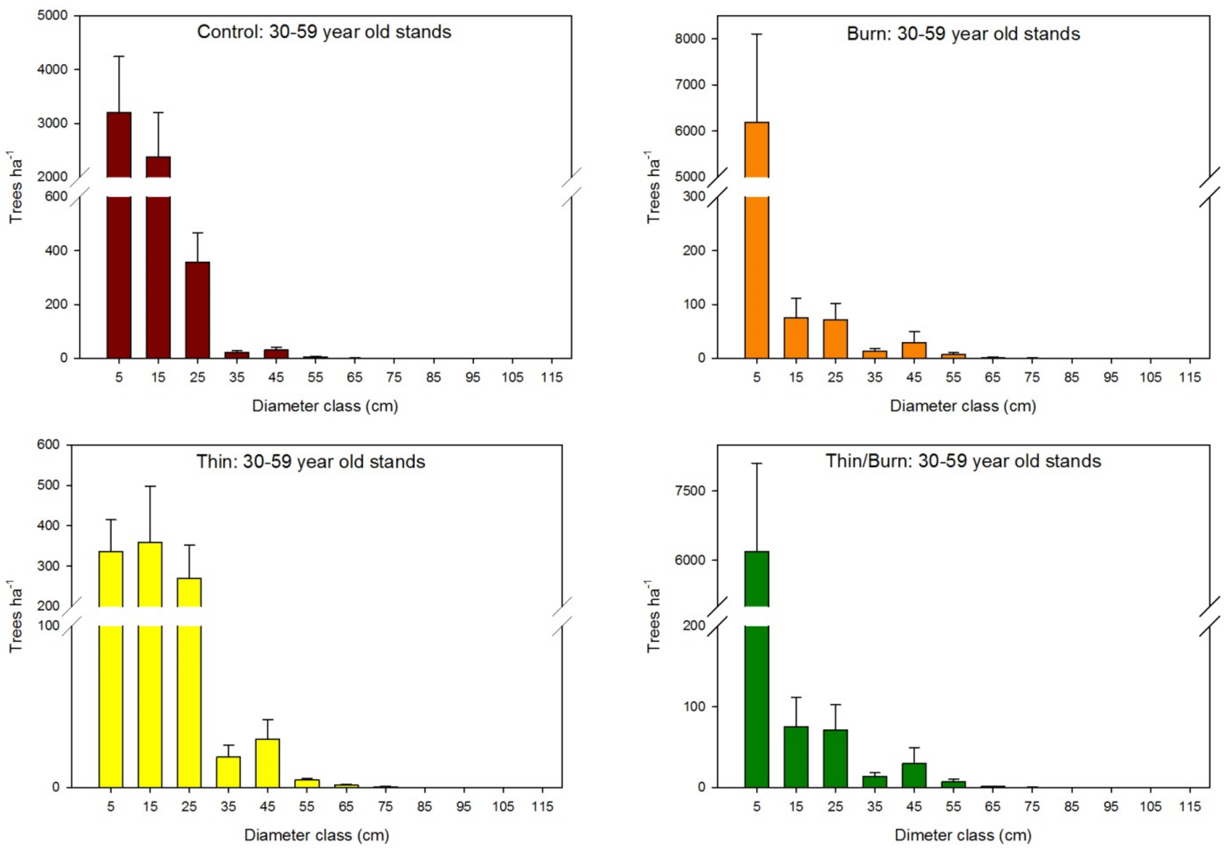

Figure 15: Year 50 simulated diameter distributions for stands with initial conditions 30-59 years old at Fort Benning by treatment. Note that the y-axis varies by treatment.

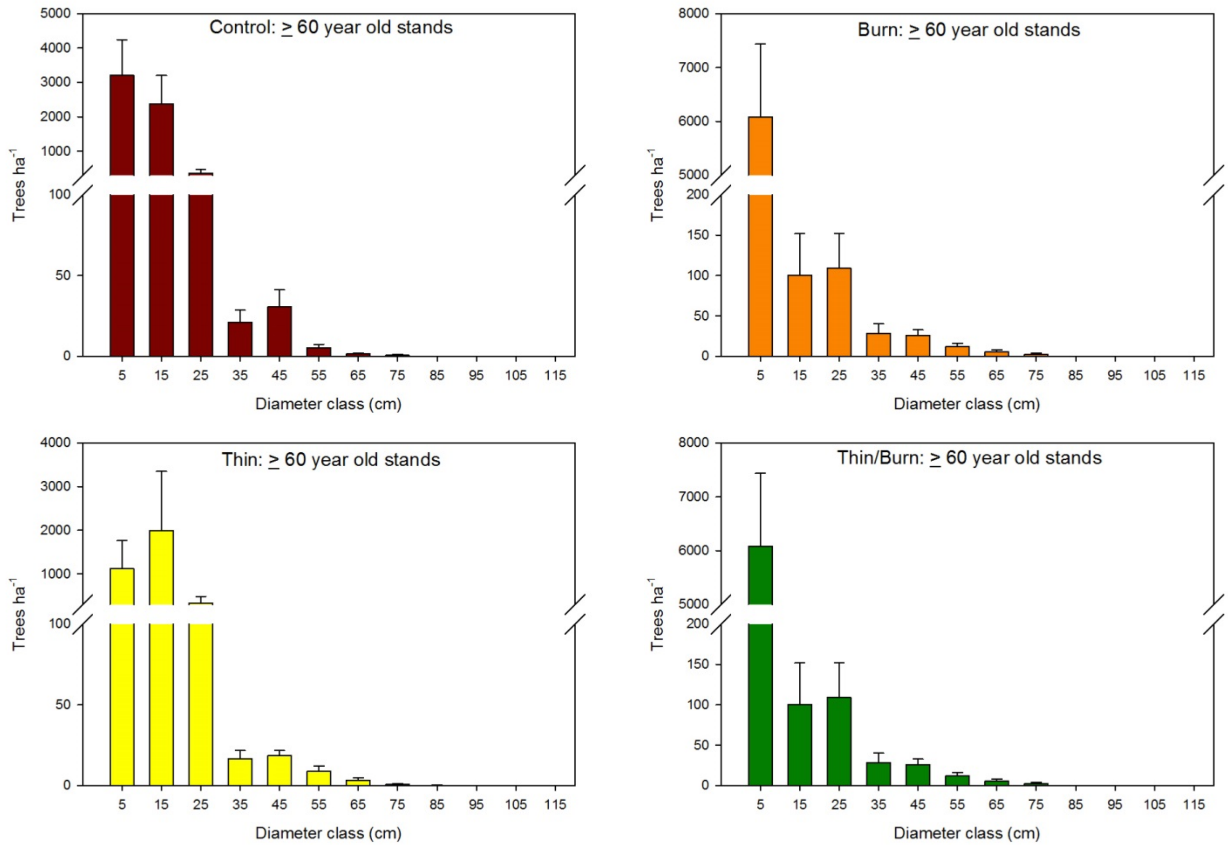

Figure 16: Year 50 simulated diameter distributions for stands with initial conditions 60 years old or greater at Fort Benning by treatment. Note that the y-axis varies by treatment.

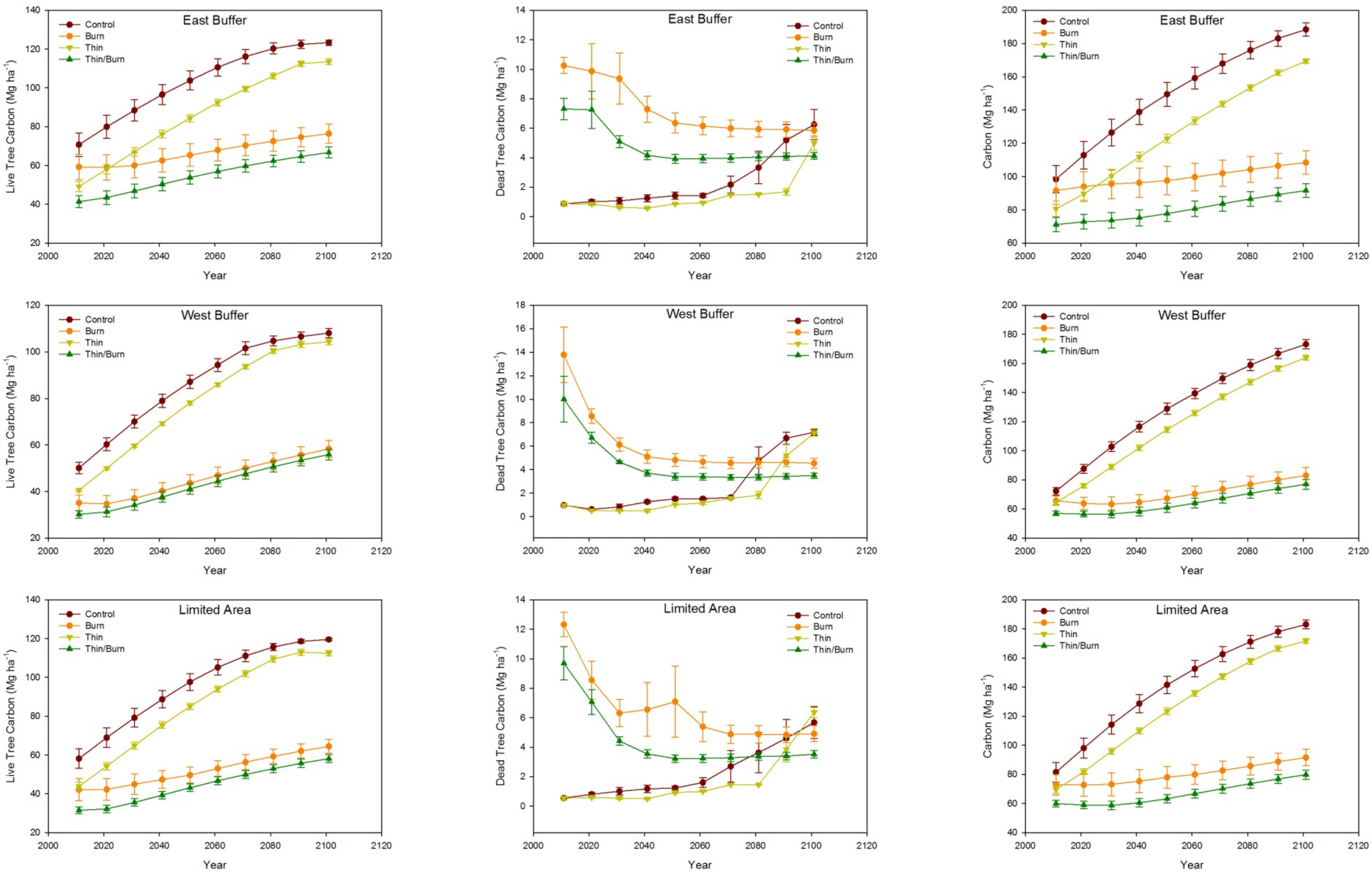

Figure 17: Live tree carbon (first column), carbon in snags (second column), and total stand carbon (third column) at Camp Navajo. Note the y-axis scale varies among graphs.

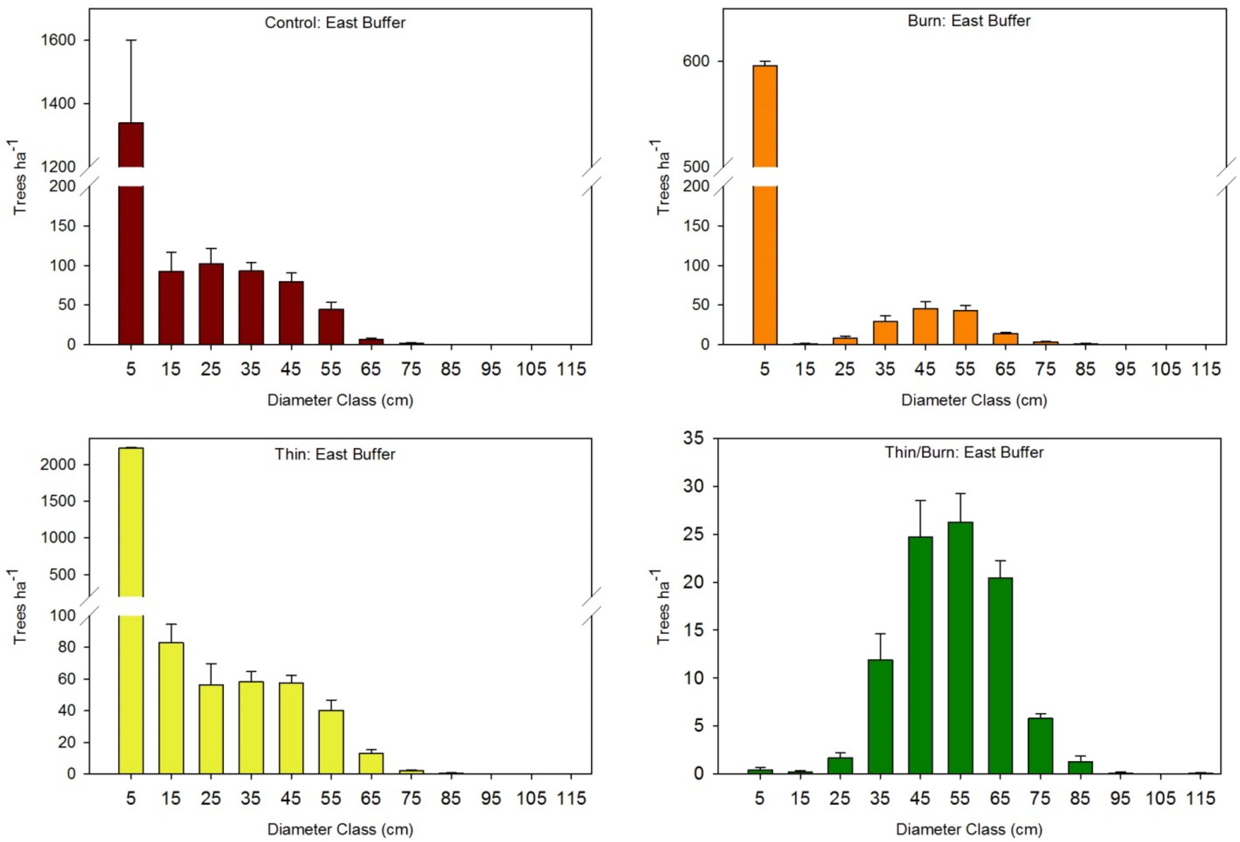

Figure 18: Year 50 simulated diameter distributions for East Buffer stands at Camp Navajo by treatment. Note that the y-axis varies by treatment.

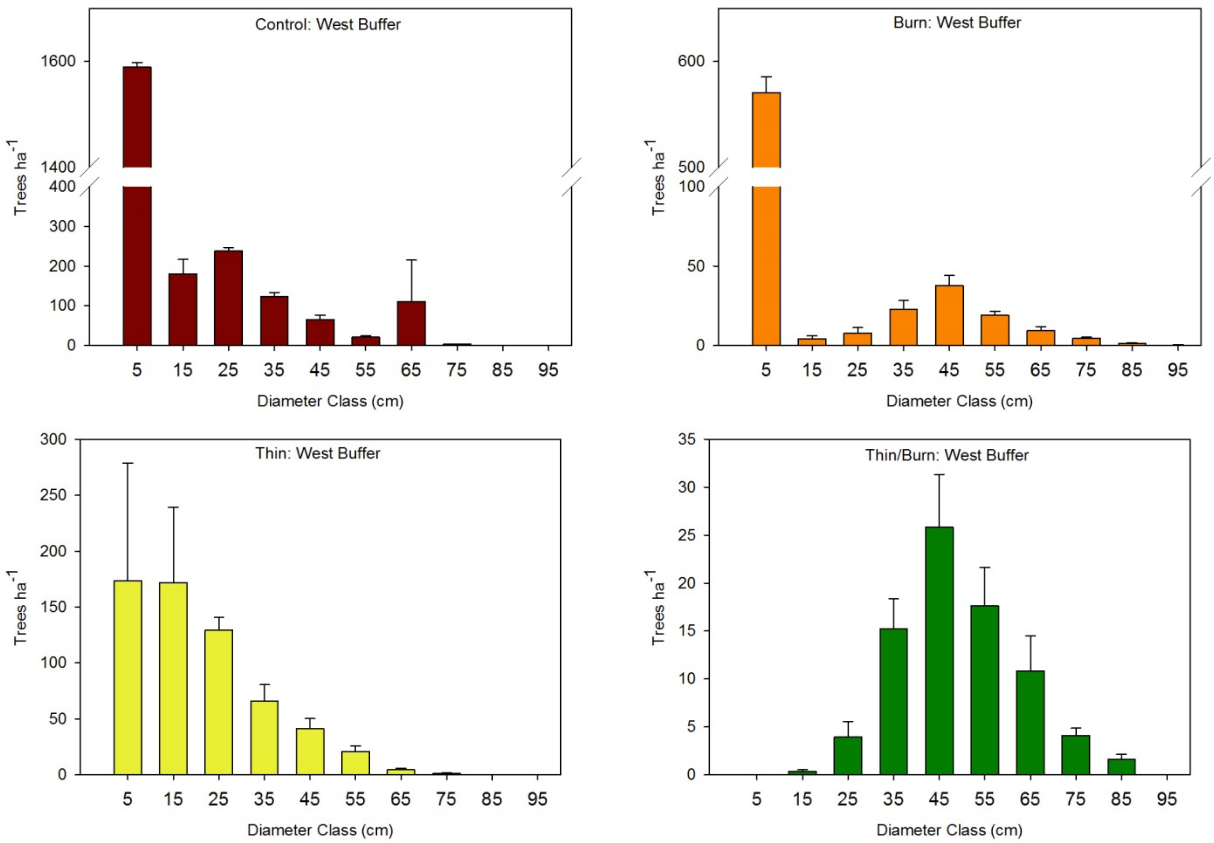

Figure 19: Year 50 simulated diameter distributions for West Buffer stands at Camp Navajo by treatment. Note that the y-axis varies by treatment.

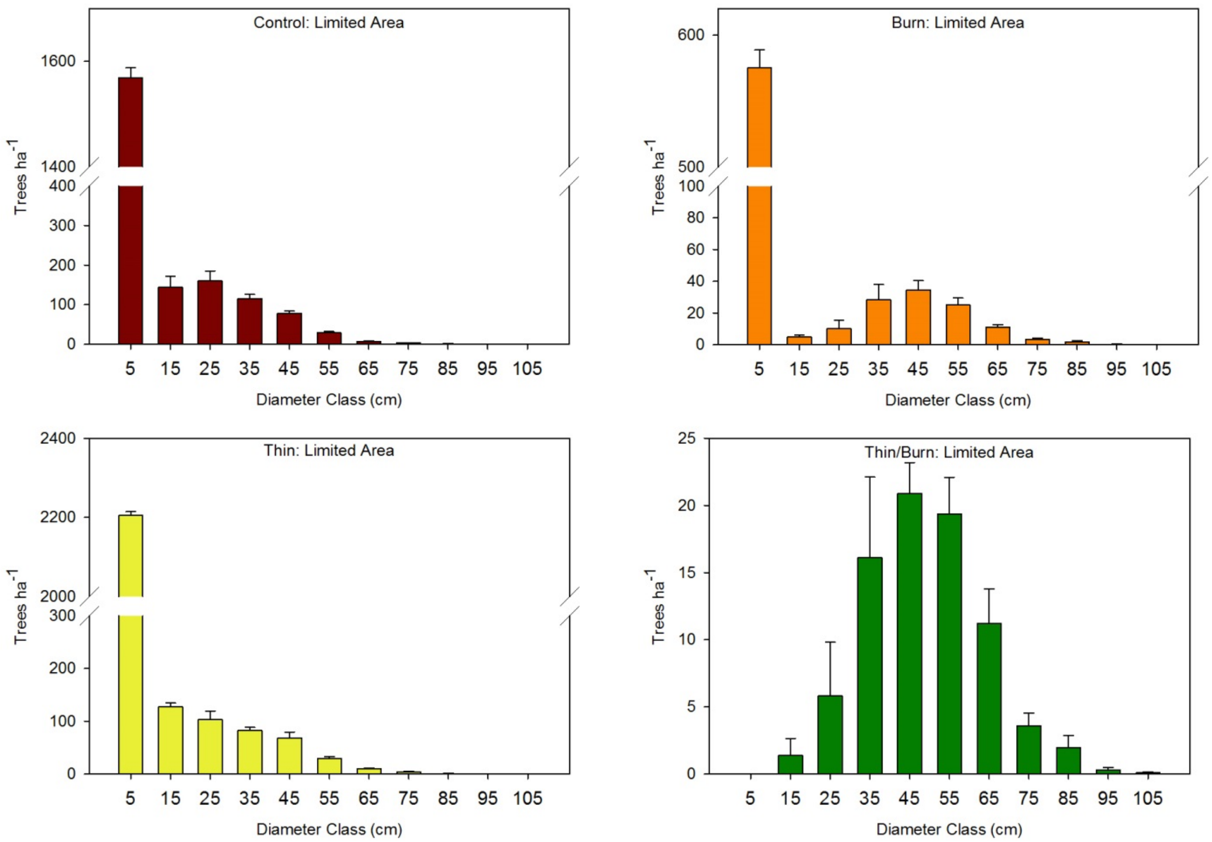

Figure 20: Year 50 simulated diameter distributions for Limited Area stands at Camp Navajo by treatment. Note that the y-axis varies by treatment.

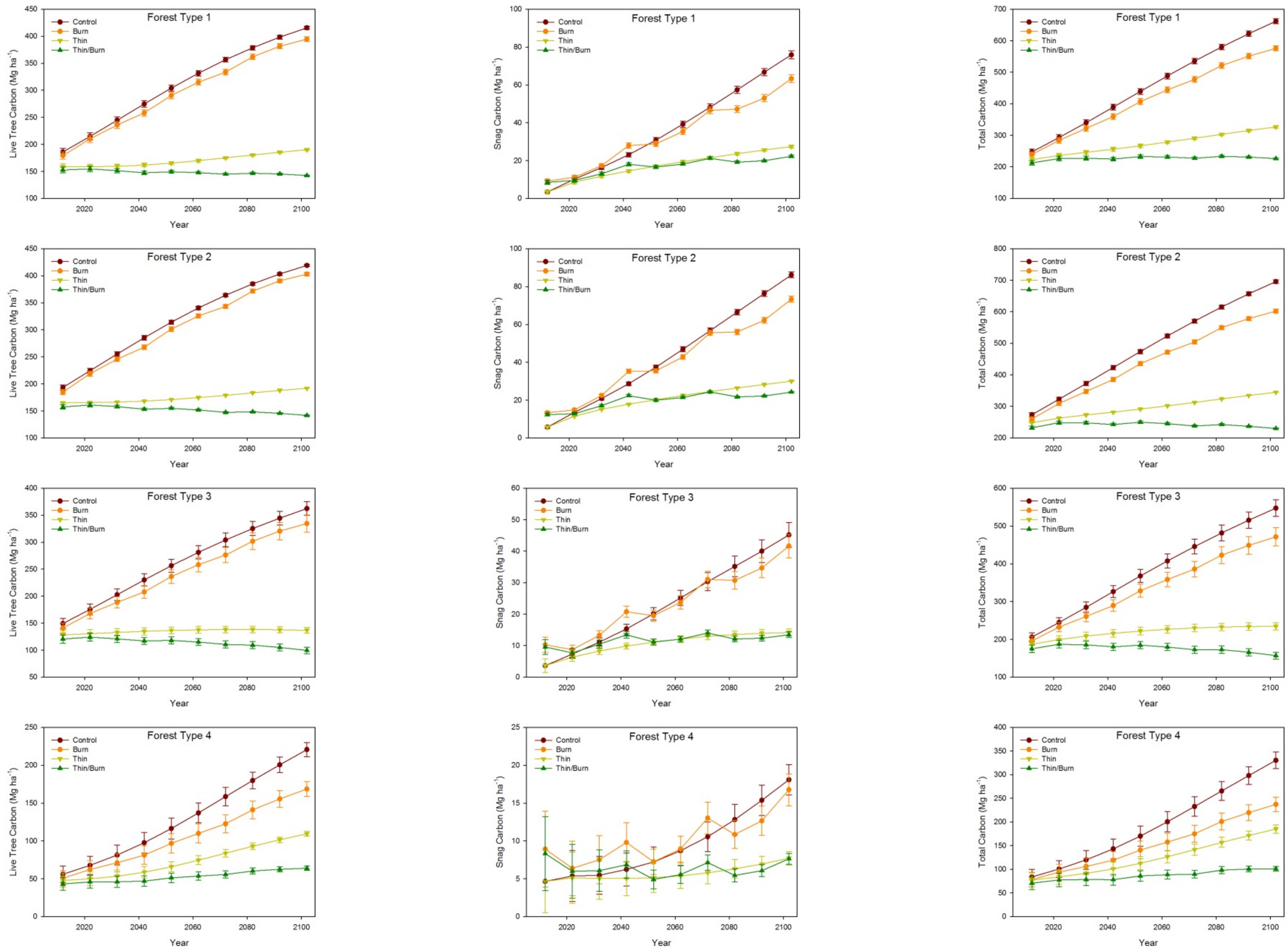

Figure 21: Live tree carbon (first column), carbon in snags (second column), and total stand carbon (third column) at Joint Base Lewis McChord. Note the y-axis scale varies among graphs.

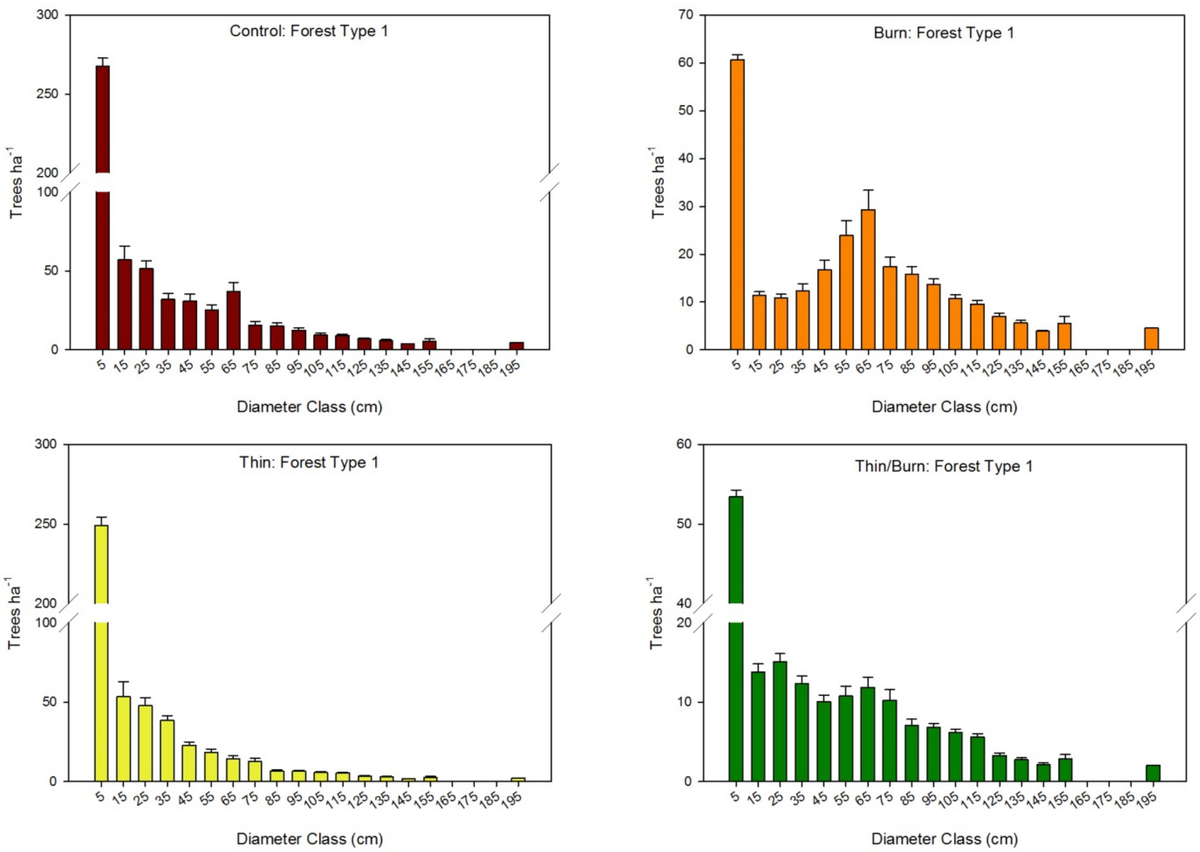

Figure 22: Year 50 simulated diameter distributions for Forest Type 1 stands at Joint Base Lewis McChord by treatment. Note that the y-axis varies by treatment.

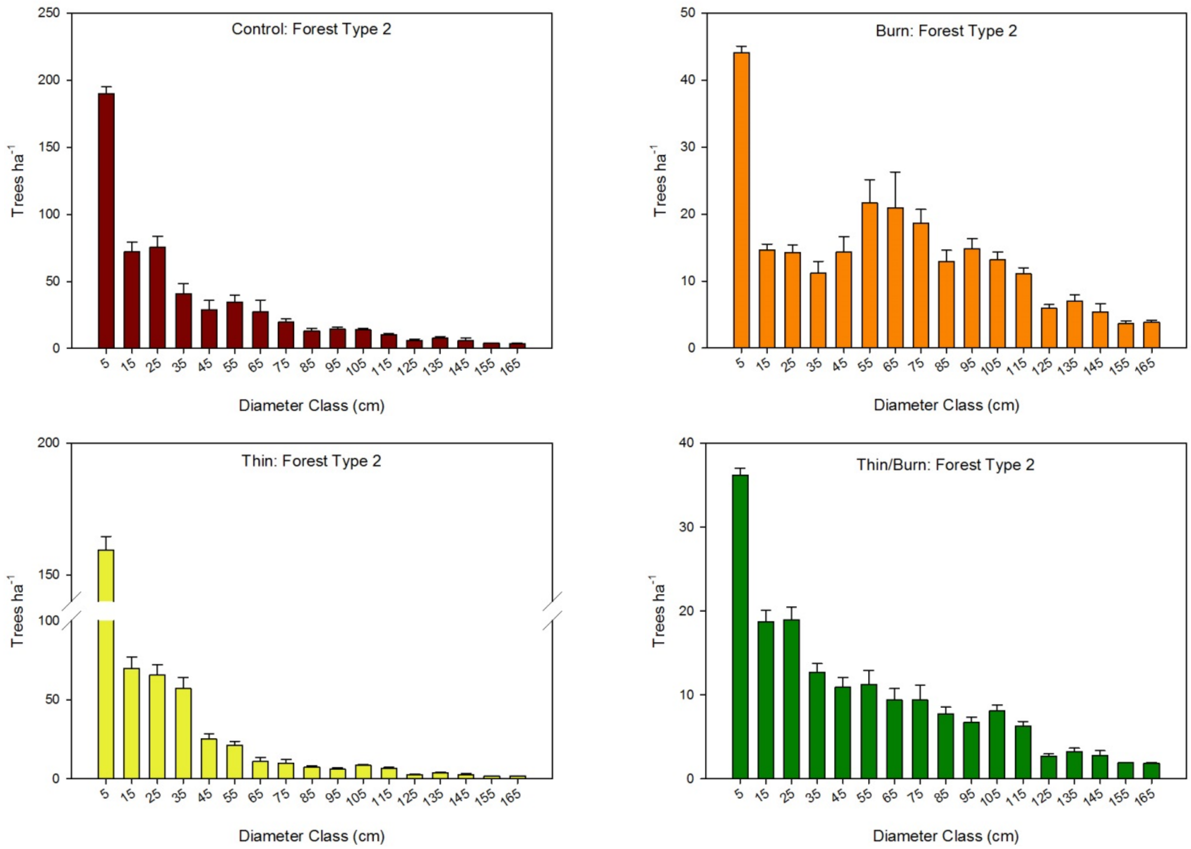

Figure 23: Year 50 simulated diameter distributions for Forest Type 2 stands at Joint Base Lewis McChord by treatment. Note that the y-axis varies by treatment.

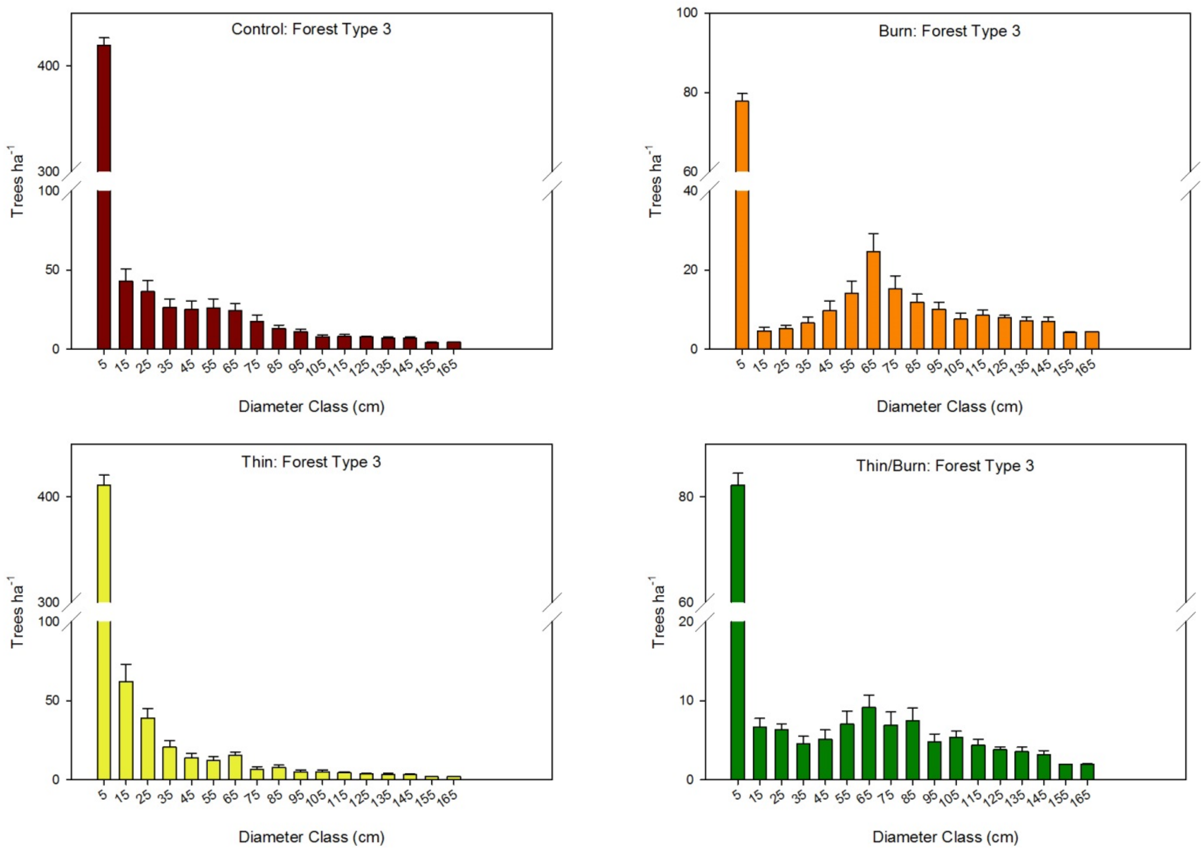

Figure 24: Year 50 simulated diameter distributions for Forest Type 3 stands at Joint Base Lewis McChord by treatment. Note that the y-axis varies by treatment.

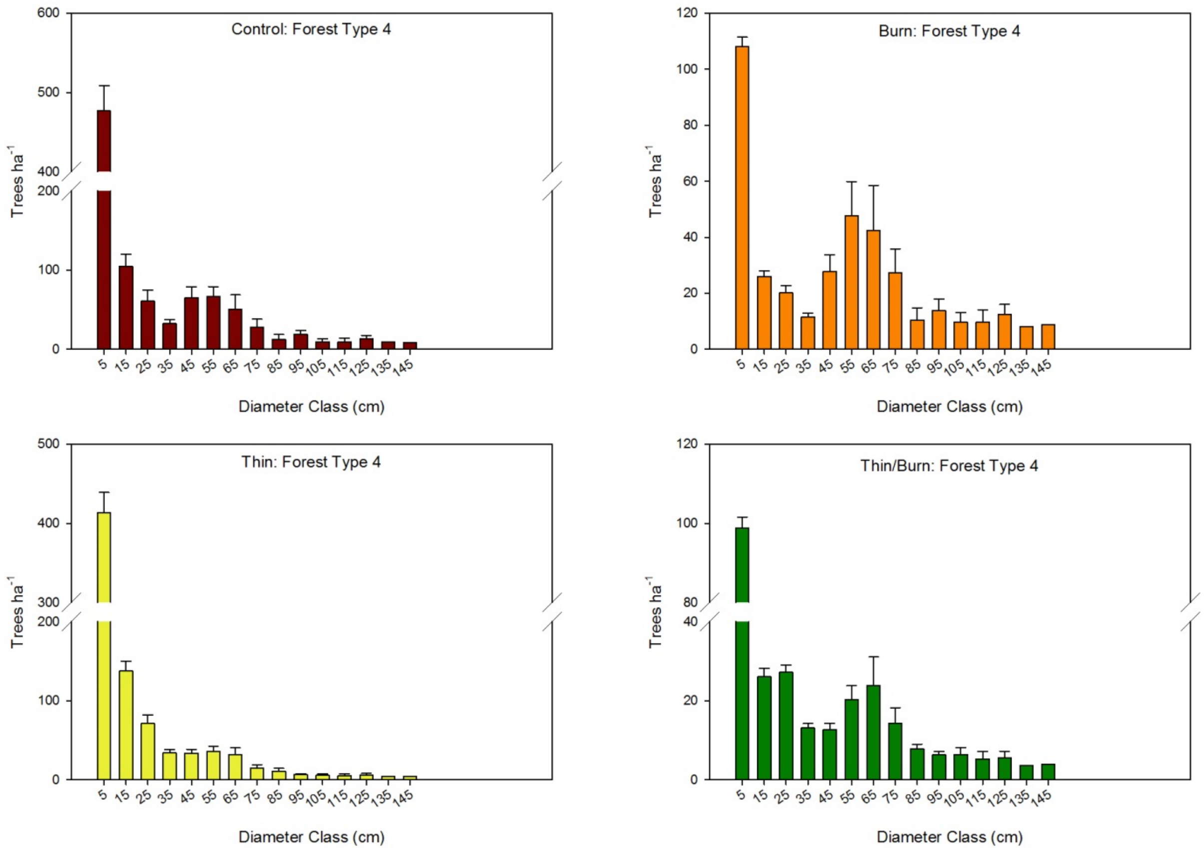

Figure 25: Year 50 simulated diameter distributions for Forest Type 4 stands at Joint Base Lewis McChord by treatment. Note that the y-axis varies by treatment.

FVS Simulation Harvest and Emissions

Thinning simulations at FB were conditionally set to occur when SDI reached 250 based on recommendations for Red-cockaded woodpecker habitat. This resulted in thinning occurring 30 and 50 years following the initiation of simulations in the thin-only and thin and burn, respectively. The C removed during thinning was highest in the young stands (Table 4). Regular prescribed burning in the thin and burn treatment influenced both the timing and C removed during thinning (Table 4). Prescribed fires were implemented every three years, consistent with the current fire rotation at the installation. Emissions in young stands were larger and more variable than mid or old stands (Figure 26). Following simulation initiation, fire emissions leveled off at approximately 7 Mg C ha^{-1} per decade (~2.2-2.5 Mg C ha^{-1} for each prescribed burn) in mid and old stands (Figure 26).

Table 4: Mean and standard error carbon removed during thinning by treatment for each stratum at Fort Benning. Young stands are < 10 years old, Mid stands are 30-59 years old, and Old stands are > 60 years old.

Treatment (thin year)	Young (se) Mg C ha^{-1}	Mid (se) Mg C ha^{-1}	Old (se) Mg C ha^{-1}
Thin-only (2041)	24.4 (5.1)	4.1 (1.5)	6.6 (2.8)
Thin/Burn (2061)	14.3 (6.0)	0.05 (0.05)	4.2 (3.4)

Thinning treatments at CN were implemented during the first year of the simulation based on the high degree of stand-replacing fire risk. Since the same thinning prescription was applied to the thin-only and thin and burn treatments, C removed was consistent across treatments with each stratum (Table 5). The lower amount of C thinned from the West Buffer reflects the fuels reduction treatments that have already been implemented in part of the stratum. Prescribed fires were implemented every ten years in each stratum. Patterns of emissions between treatments were consistent across strata (Figure 27). Burn-only treatments were consistently higher from the second prescribed fire onward. Thin and burn treatments were higher during the first simulated burn because of the added fuels from thinning implementation, but dropped to an approximate average of 3 Mg C ha^{-1} by the middle of the simulation period.

Table 5: Mean and standard error carbon removed during thinning by treatment at Camp Navajo. Harvested carbon values are presented by stratum for the East Buffer (EB), West Buffer (WB), and Limited Area (LA).

Treatment (thin year)	EB (se) Mg C ha^{-1}	WB (se) Mg C ha^{-1}	LA (se) Mg C ha^{-1}
Thin-only (2011)	15.7 (3.8)	6.4 (1.9)	10.2 (4.6)
Thin/Burn (2011)	15.7 (3.8)	6.4 (1.9)	10.2 (4.6)

Thinning treatments at JBLM were based on producing a sustainable yield of harvested wood volume and reflect current harvesting practices of removing 15% of the basal area with each entry. Harvests were simulated every ten years. Following the initial thinning entry, the thin-only treatments had consistently larger carbon removals than the thin and burn because of the effects of regular prescribed burning on regeneration and mortality (Table 6). Prescribed fires were implemented every 15 years at JBLM. Prescribed fire emissions varied by forest type, but generally increased by the end of the simulation period for the burn-only treatment (Figure 28). Emissions in the thin and burn treatment oscillated between burns as a function of thinning implementation, with years that included thinning having larger emissions than interim years (Figure 28).

Table 6: Mean and standard error carbon (Mg C ha-1) removed during thinning by treatment at Joint Base Lewis McChord.

	Forest Type 1		Forest Type 2		Forest Type 3		Forest Type 4	
Year	Thin	Thin/Burn	Thin	Thin/Burn	Thin	Thin/Burn	Thin	Thin/Burn
2012	24.0 (0.9)	24.0 (0.9)	24.8 (0.1)	24.8 (0.6)	18.6 (1.2)	18.6 (1.2)	6.3 (1.4)	6.3 (1.4)
2022	23.9 (0.7)	23.6 (0.7)	25.0 (0.1)	24.5 (0.5)	19.3 (1.2)	18.7 (1.1)	6.6 (1.3)	6.0 (1.2)
2032	24.0 (0.6)	23.2 (0.6)	25.0 (0.1)	24.1 (0.4)	19.7 (1.1)	18.6 (1.1)	6.8 (1.2)	6.0 (1.1)
2042	24.2 (0.5)	23.3 (0.5)	25.0 (0.1)	24.1 (0.3)	20.1 (1.1)	18.8 (1.0)	7.4 (1.1)	6.4 (0.9)
2052	24.9 (0.4)	23.1 (0.4)	25.3 (0.1)	23.7 (0.3)	20.5 (1.1)	18.5 (1.0)	8.3 (1.0)	6.6 (0.8)
2062	25.6 (0.3)	22.9 (0.3)	26.0 (0.1)	23.4 (0.3)	20.7 (1.1)	18.1 (1.0)	9.3 (0.9)	6.9 (0.7)
2072	26.5 (0.2)	23.1 (0.3)	26.7 (0.1)	23.4 (0.2)	20.7 (1.1)	17.3 (1.0)	10.6 (0.8)	7.6 (0.6)
2082	27.3 (0.2)	23.0 (0.2)	27.5 (0.1)	23.1 (0.2)	20.7 (1.1)	17.3 (1.0)	11.9 (0.7)	8.0 (0.5)
2092	28.1 (0.2)	22.8 (0.2)	28.3 (0.0)	22.8 (0.2)	20.6 (1.0)	16.7 (1.0)	13.2 (0.6)	8.4 (0.5)
2102	28.8 (0.1)	23.0 (0.1)	29.0 (0.0)	22.8 (0.1)	20.3 (0.9)	16.2 (0.9)	14.4 (0.6)	9.0 (0.4)

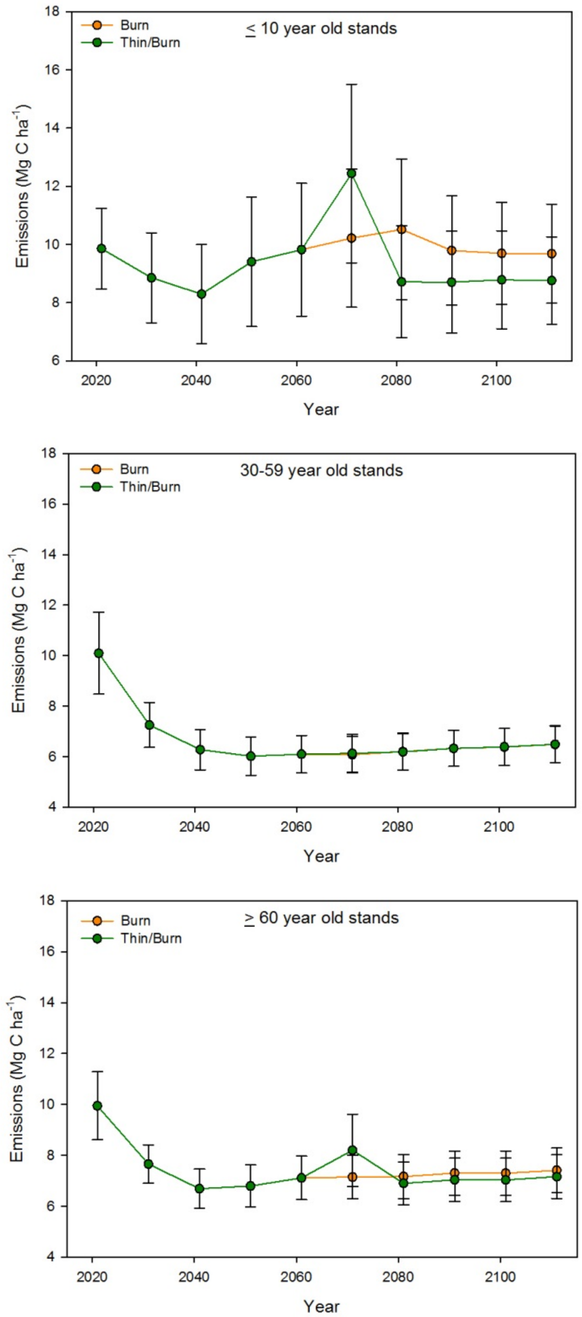

Figure 26: Prescribed fire emissions by treatment for each stratum at Fort Benning. Note the y-axis scale varies by stand type.

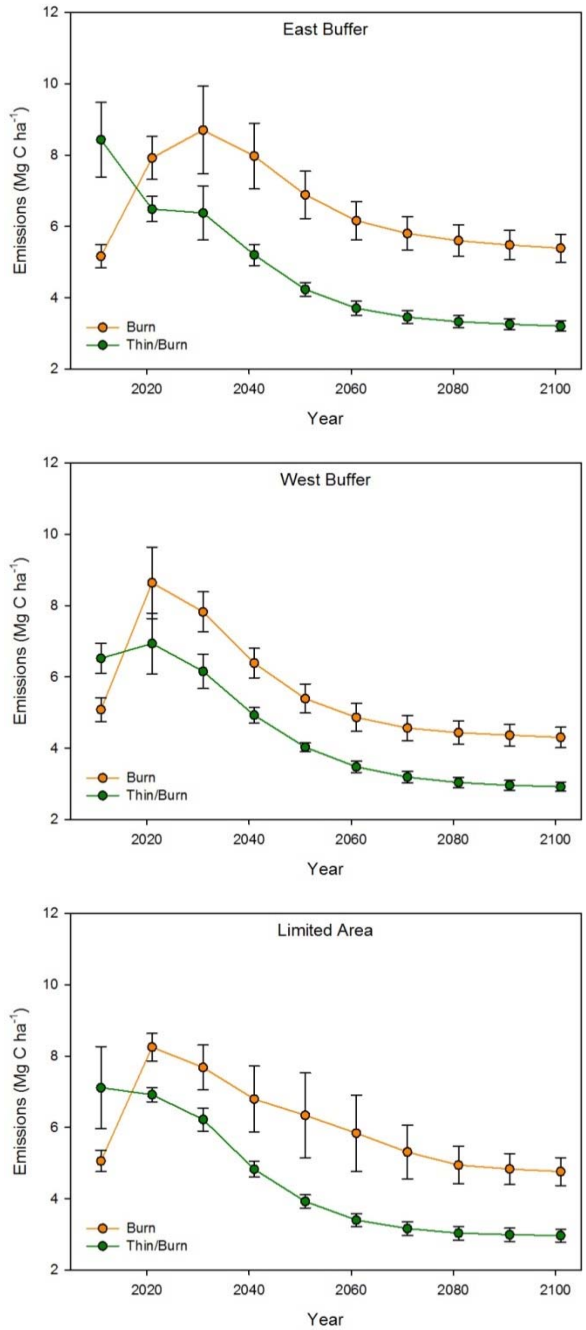

Figure 27: Prescribed fire emissions by treatment for each stand type at Camp Navajo.

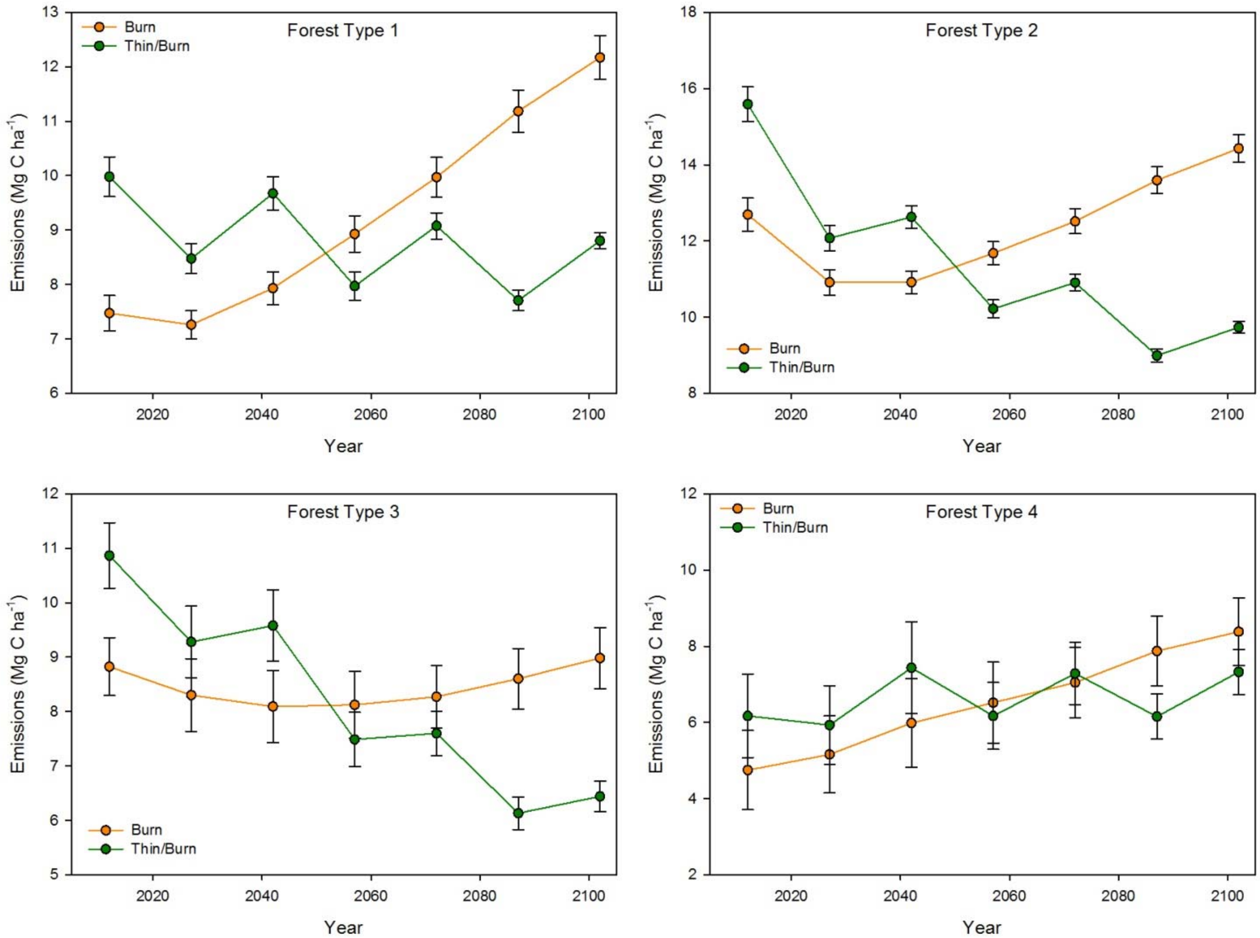

Figure 28: Prescribed fire emissions by treatment for each forest type at Joint Base Lewis McChord. Note the y-axis scale varies by forest type.

LANDIS-II Simulations – Fort Benning

To implement LANDIS-II simulations, FB was gridded using a 200 m grid divided into four ecoregions as a function of soil type (Figure 29). The grid size was selected based on the average stand size of four hectares. Stand data collected as part of this research and inventory data from resource managers at the installation were used to assign forest types and species age cohorts to each grid cell (Figure 30). For example, longleaf forests were assigned a greater mix of species cohorts in a given grid cell designated as longleaf forest than a grid cell designated as longleaf plantation. Grid cells that fell within the range fans were excluded due to a lack of forest inventory data for parameterization. Species-specific parameters were obtained from the literature when available or estimated using parameter values from different species within the same genus (Appendix I).

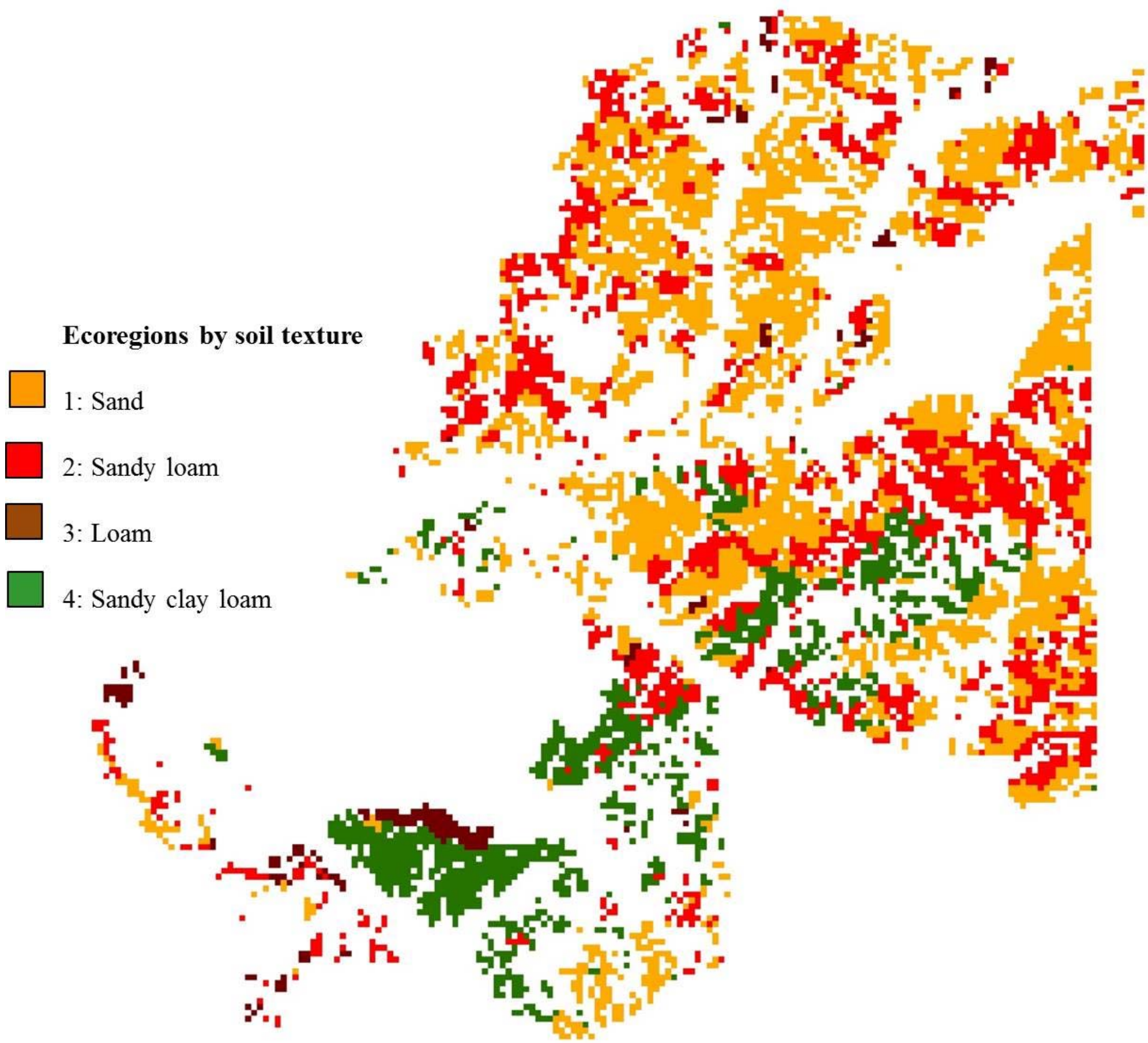

Figure 29: LANDIS-II ecoregions identified as a function of soil type from the NRCS SSURGO soils database for Fort Benning.

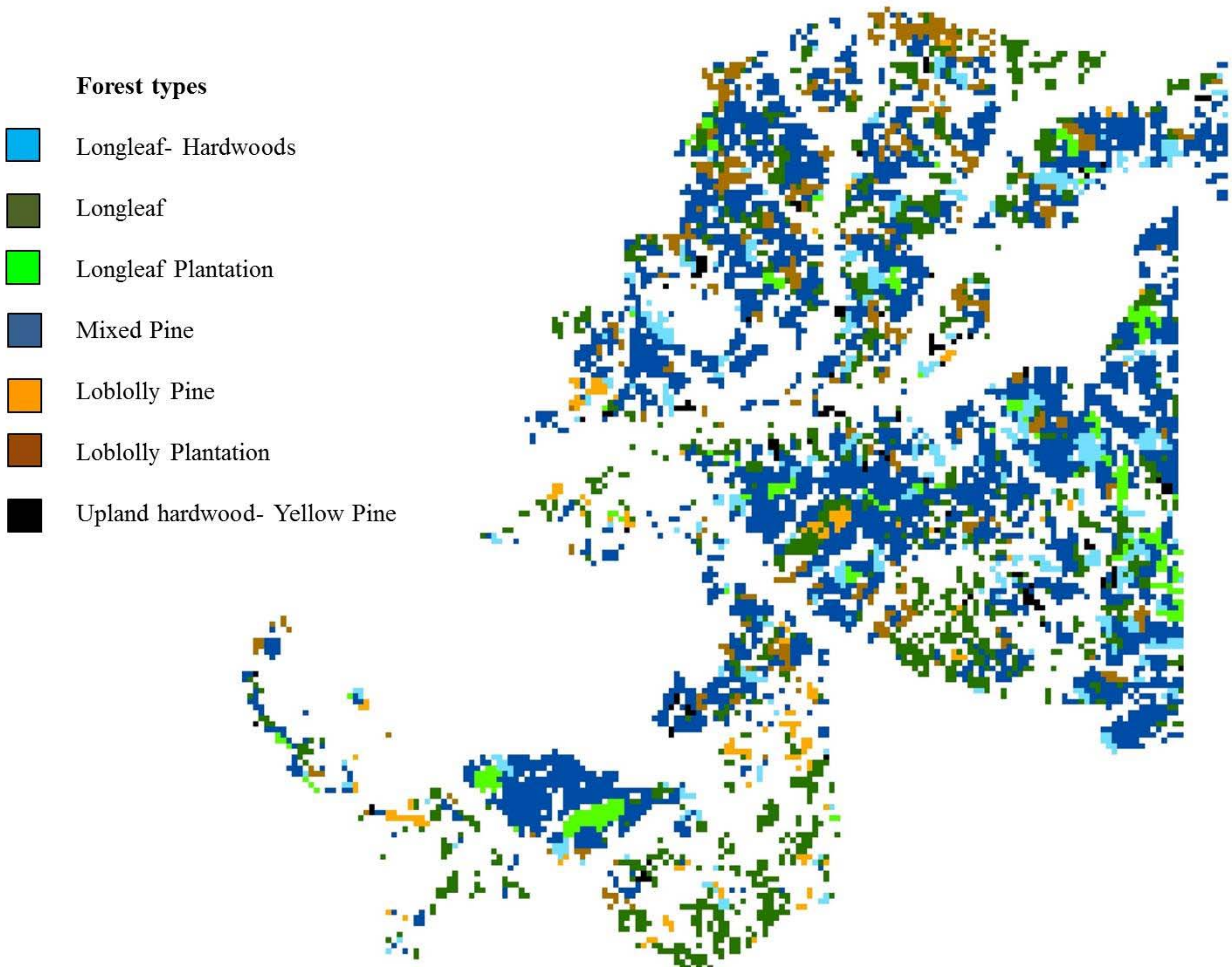

Figure 30: Forest types assigned to 200 m grid cells based on inventory data for Fort Benning.

Control simulations were run in LANDIS-II to characterize the C dynamics in the system in the absence of management. Net Ecosystem Exchange (NEE) was negative throughout most of the simulation period, indicating that the forest types simulated were a net C sink (Figure 31). Ecoregion 4, which occupies a relatively small area of FB did achieve zero NEE by the end of the simulation, indicating that Net Primary Productivity (NPP) was equivalent to Respiration (R) for forest types in this ecoregion (Figure 31). NEE results are presented by ecoregion because of the influence of soil properties on NPP and R. In the absence of disturbance, soil organic carbon (SOC) increased in all ecoregions; doubling in quantity in three of the four ecoregions (Figure 32). Initial aboveground C stocks ranged from 0-50 Mg C ha^{-1} for the forest types simulated (Figure 33). By year 50 of the control simulation the majority of forested areas simulated had aboveground C stocks ranging from 50-100 Mg C ha^{-1} (Figure 34). At the conclusion of the 100-year simulation, aboveground carbon stocks had increased to 75-100 Mg C ha^{-1} across the majority of the installation (Figure 35). The spatial variability in C stocks throughout the simulation period is a function of the initial species composition and age distribution within each grid cell.

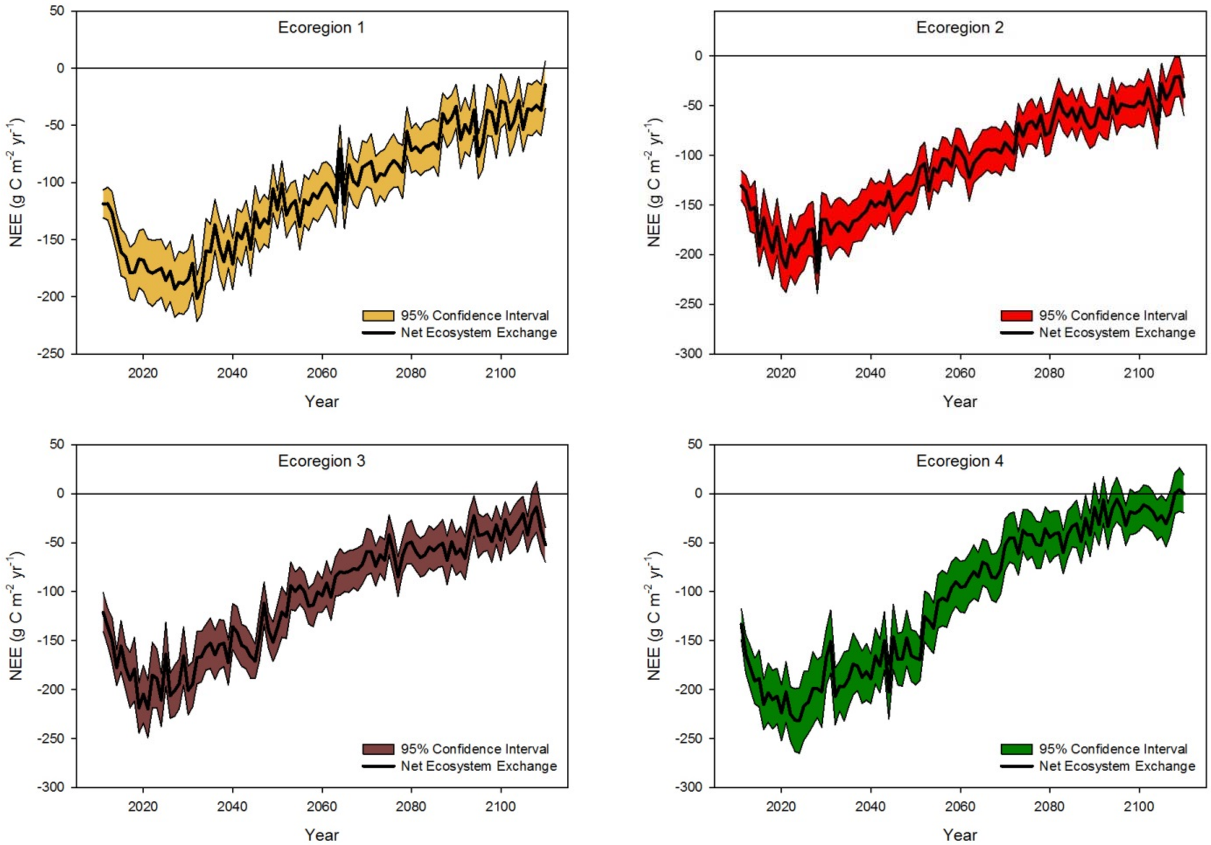

Figure 31: Net Ecosystem Exchange by ecoregion simulated in LANDIS-II with the Century Succession extension at Fort Benning for the control simulation. Negative values represent a net carbon sink. Note the y-axis varies by ecoregion.

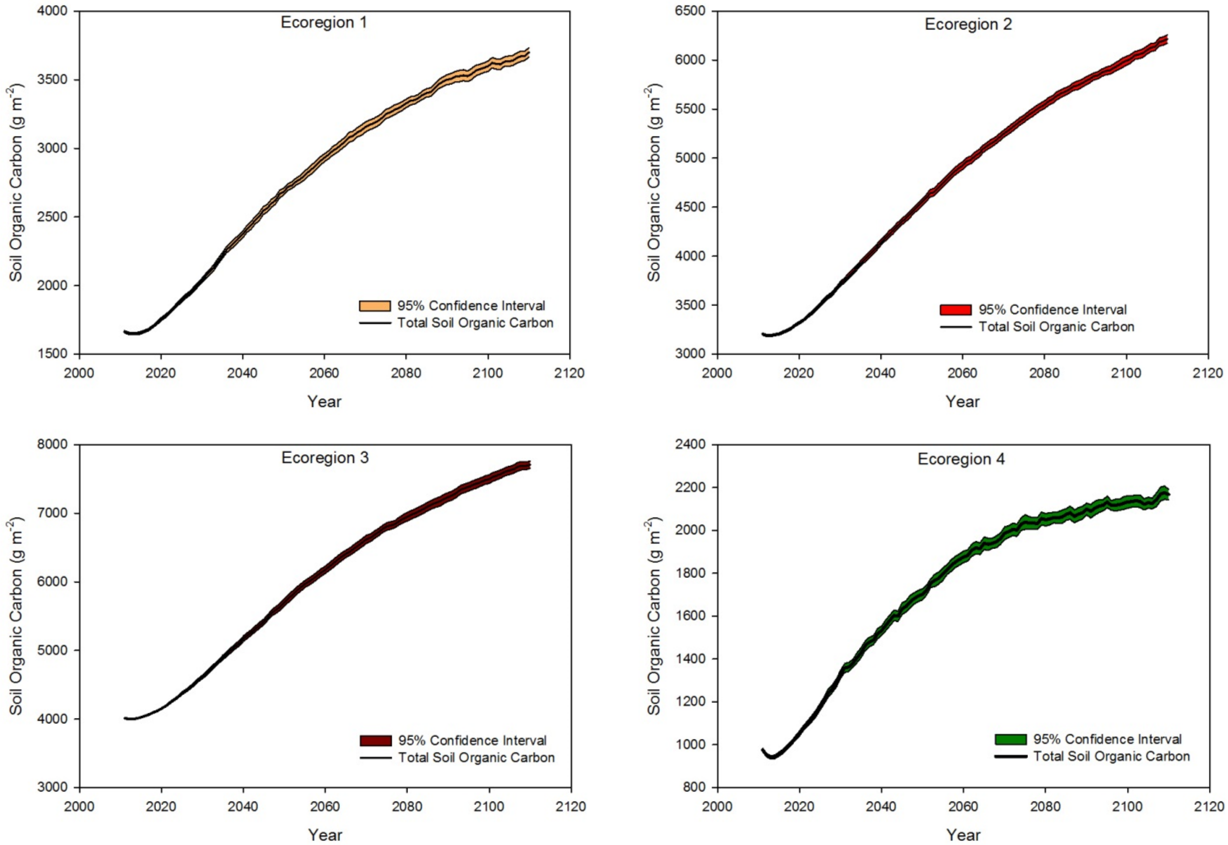

Figure 32: Soil organic carbon by ecoregion simulated in LANDIS-II with the Century Succession extension at Fort Benning for the control simulation. Note the y-axis varies by ecoregion.

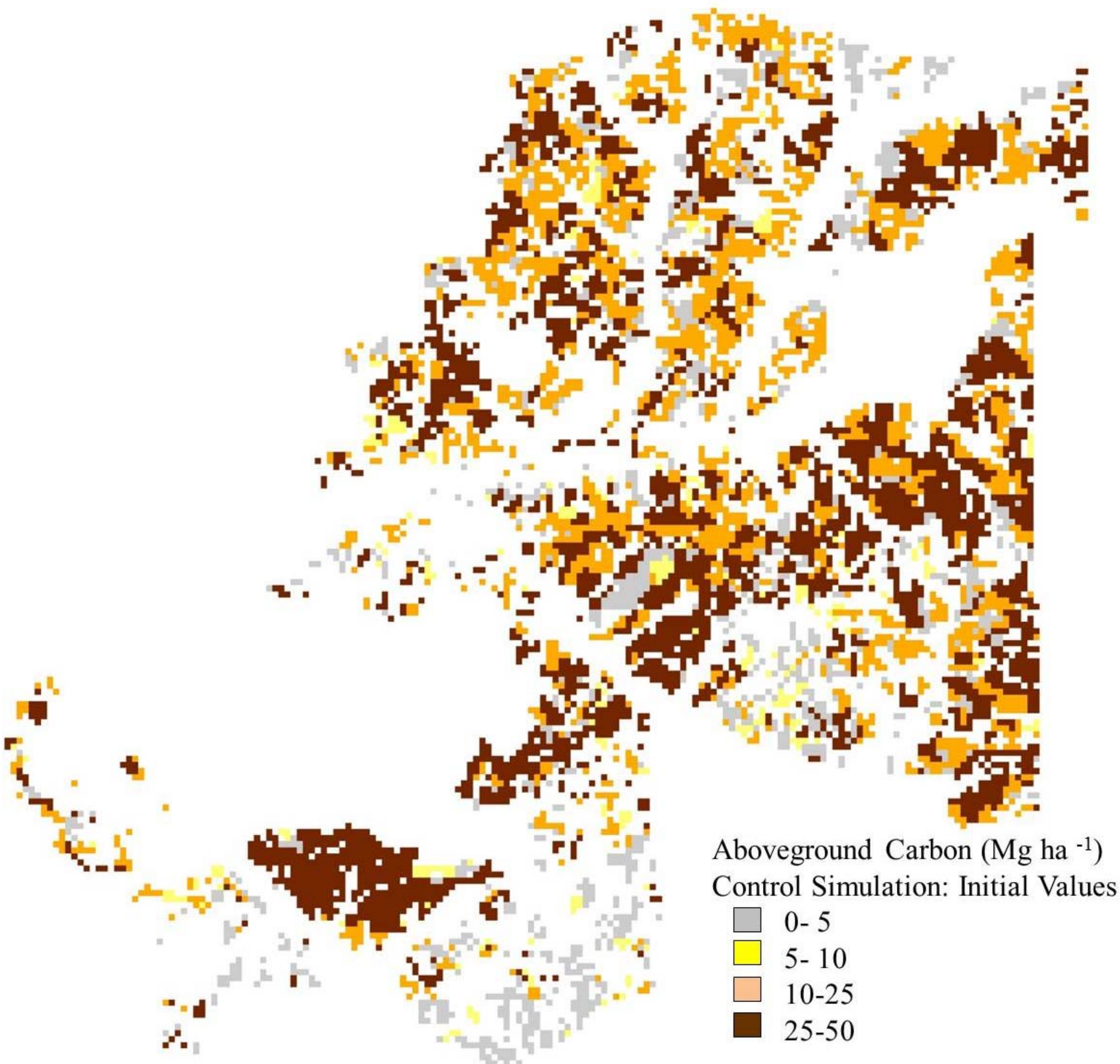

Figure 33: Initial aboveground carbon stock values simulated in LANDIS-II with the Century Succession extension at Fort Benning for the control simulation.

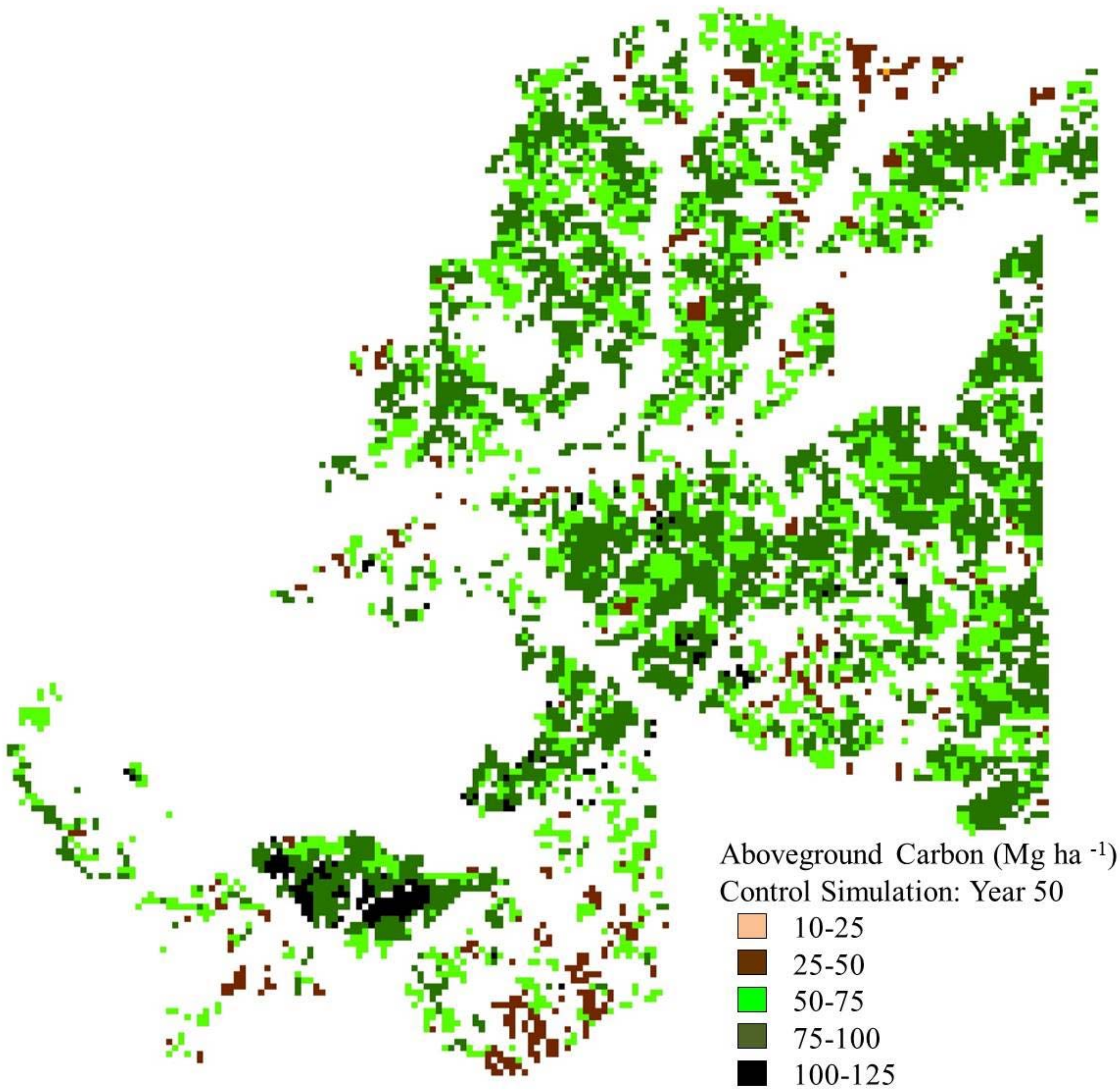

Figure 34: Year 50 aboveground carbon stock values simulated in LANDIS-II with the Century Succession extension at Fort Benning for the control simulation.

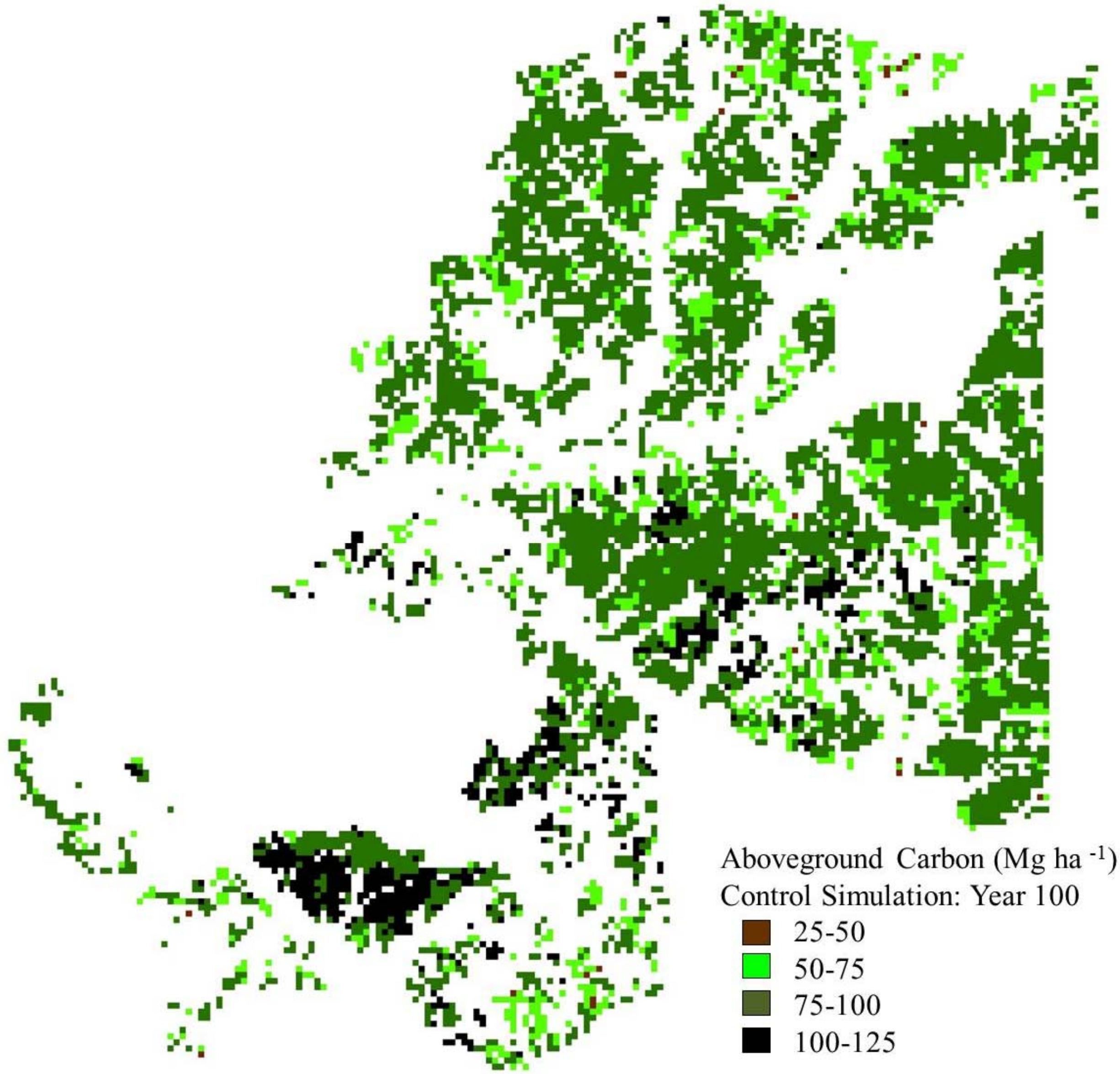

Figure 35: Year 100 aboveground carbon stock values simulated in LANDIS-II with the Century Succession extension at Fort Benning for the control simulation.

Prescribed fire simulations were run using LANDIS-II with both the Century Succession and Fire extensions. For the prescribed fire simulations, fire was simulated stochastically so that one-third of the pixels burned every year. Within a given pixel, when fire is initiated, 70% of the pixel is burned. The three year fire return interval represents current management practice at the installation. Throughout the simulation period, NEE showed similar results to the control scenario by ecoregion, with ecoregions 1-3 being net carbon sinks throughout the 100 year simulation (Figure 35). Soil organic carbon declined relative to the control simulations. This is due to a reduction in organic inputs into the soil layer because of the repeated burning (Figure 36). Across the installation, aboveground C stocks varied as a function of forest type and fire frequency (Figures 37-39). By year 50 of the simulation period, the prescribed fire simulations had a greater fraction of the land area in the 10-25 Mg C ha^{-1} range relative to the control (Figure 38). By the end of the simulation period, the prescribed fire simulation continued to maintain aboveground C in the 10-25 Mg C ha^{-1} range, while in the control simulation, all cells had surpassed this range (Figure 39). This difference is driven by fire-induced tree mortality from repeated burning.

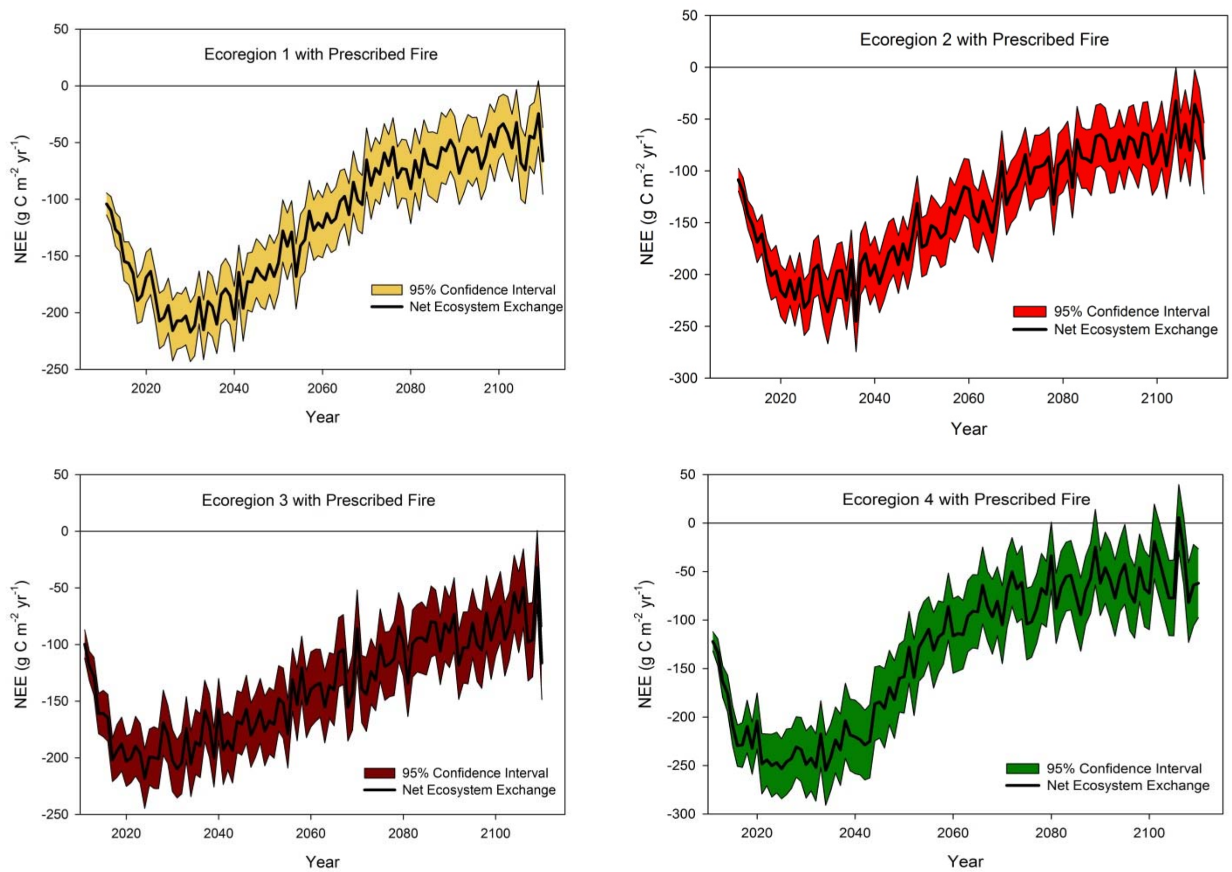

Figure 36: Net Ecosystem Exchange by ecoregion simulated in LANDIS-II with the Century Succession extension at Fort Benning for the prescribed fire simulation. Negative values represent a net carbon sink. Note the y-axis varies by ecoregion.

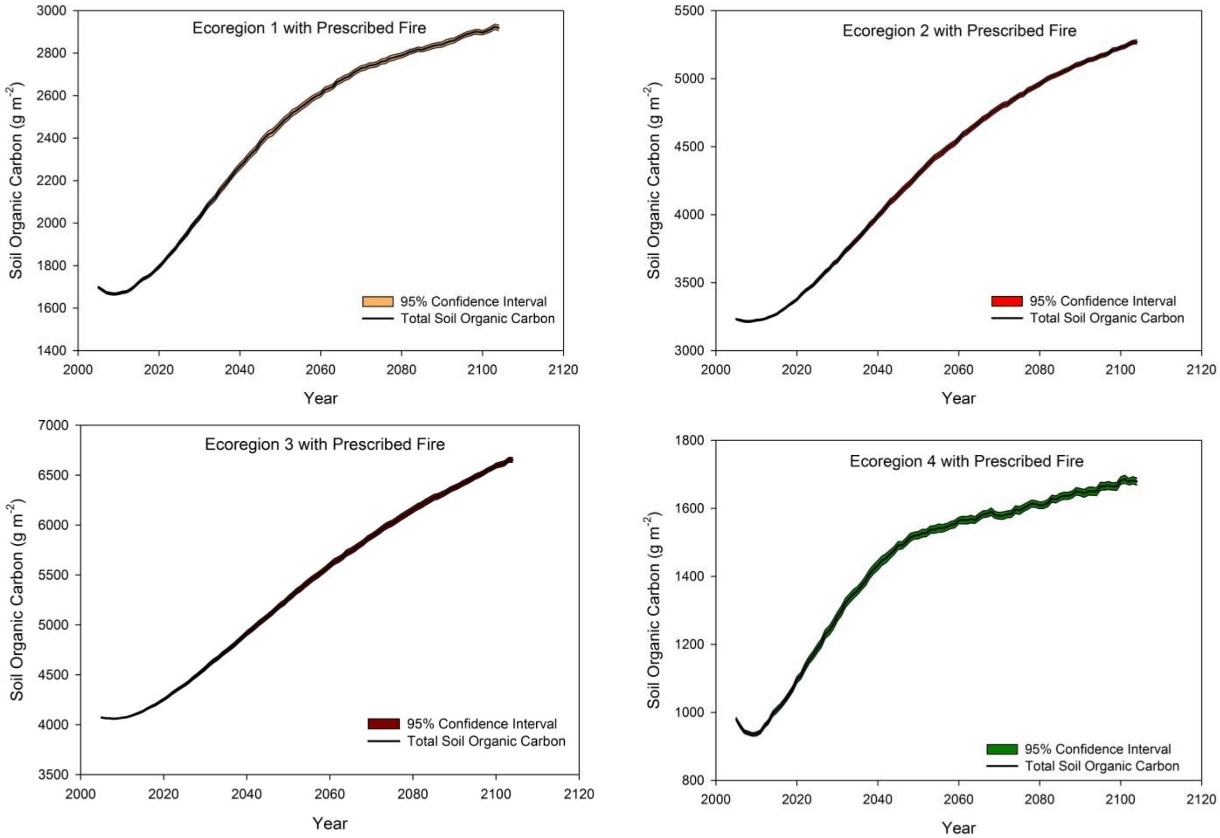

Figure 37: Soil organic carbon by ecoregion simulated in LANDIS-II with the Century Succession extension at Fort Benning for the prescribed fire simulation. Note the y-axis varies by ecoregion.

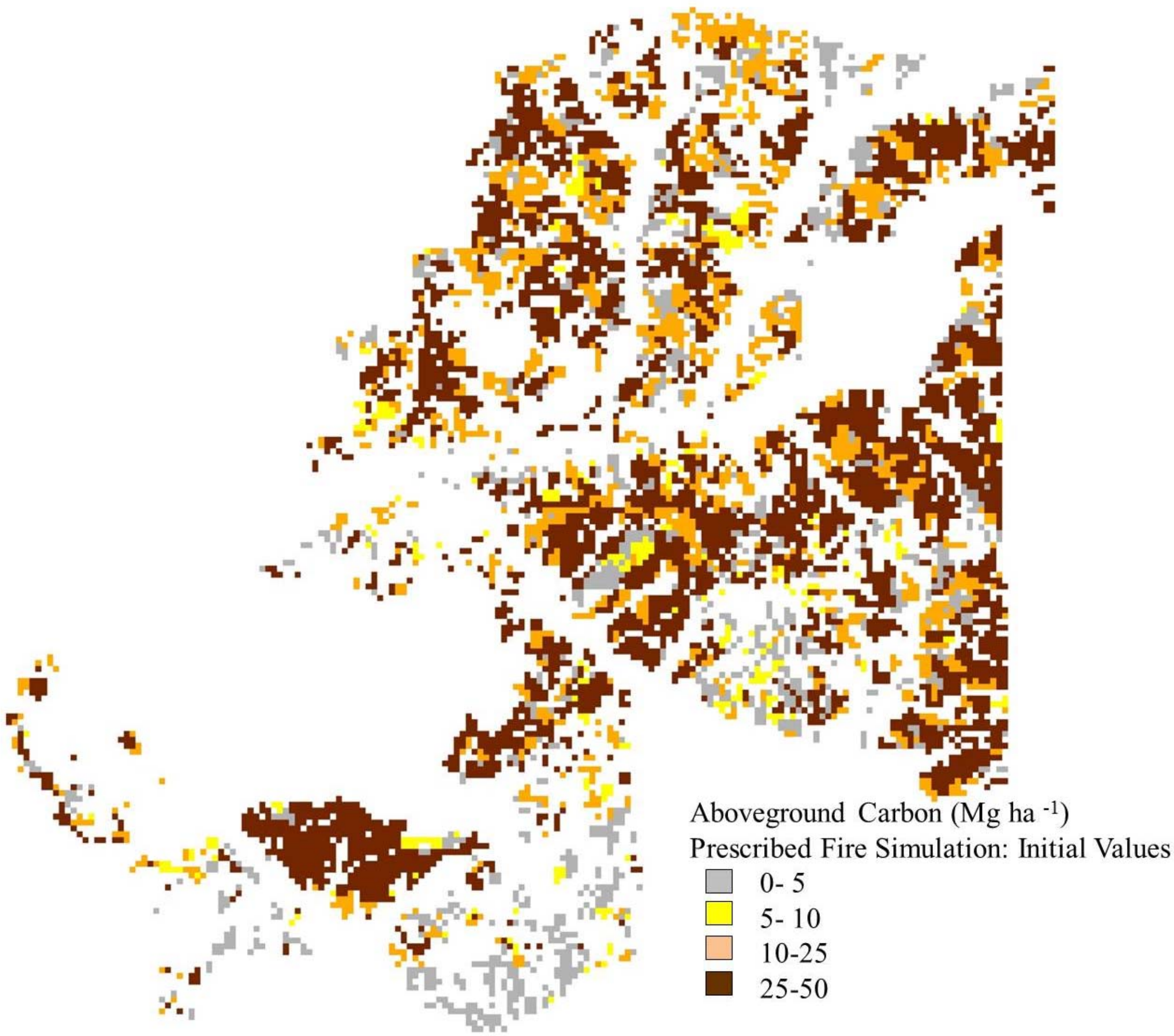

Figure 38: Initial aboveground carbon stock values simulated in LANDIS-II with the Century Succession extension at Fort Benning for the prescribed fire simulation.

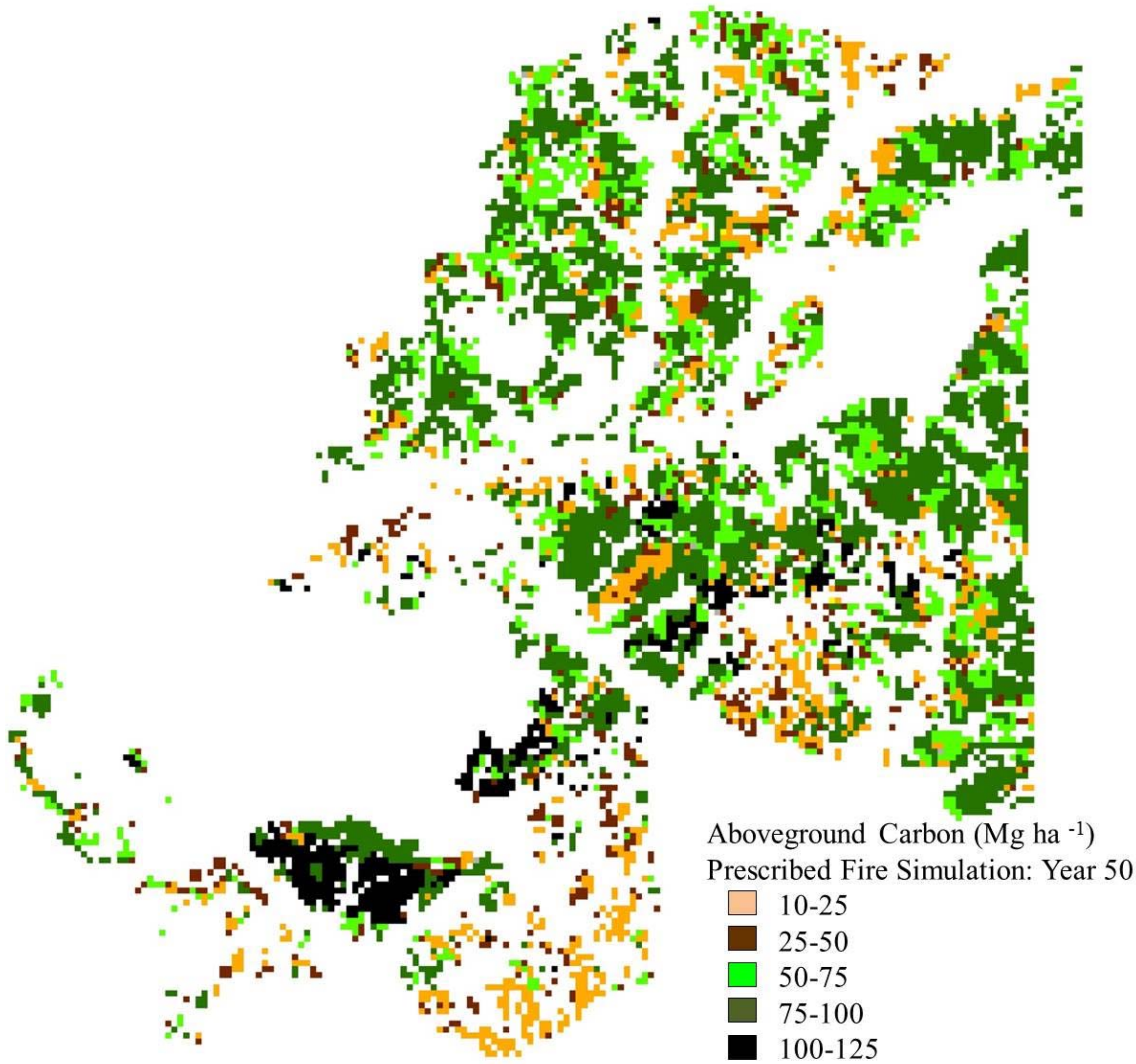

Figure 39: Year 50 aboveground carbon stock values simulated in LANDIS-II with the Century Succession extension at Fort Benning for the prescribed fire simulation.

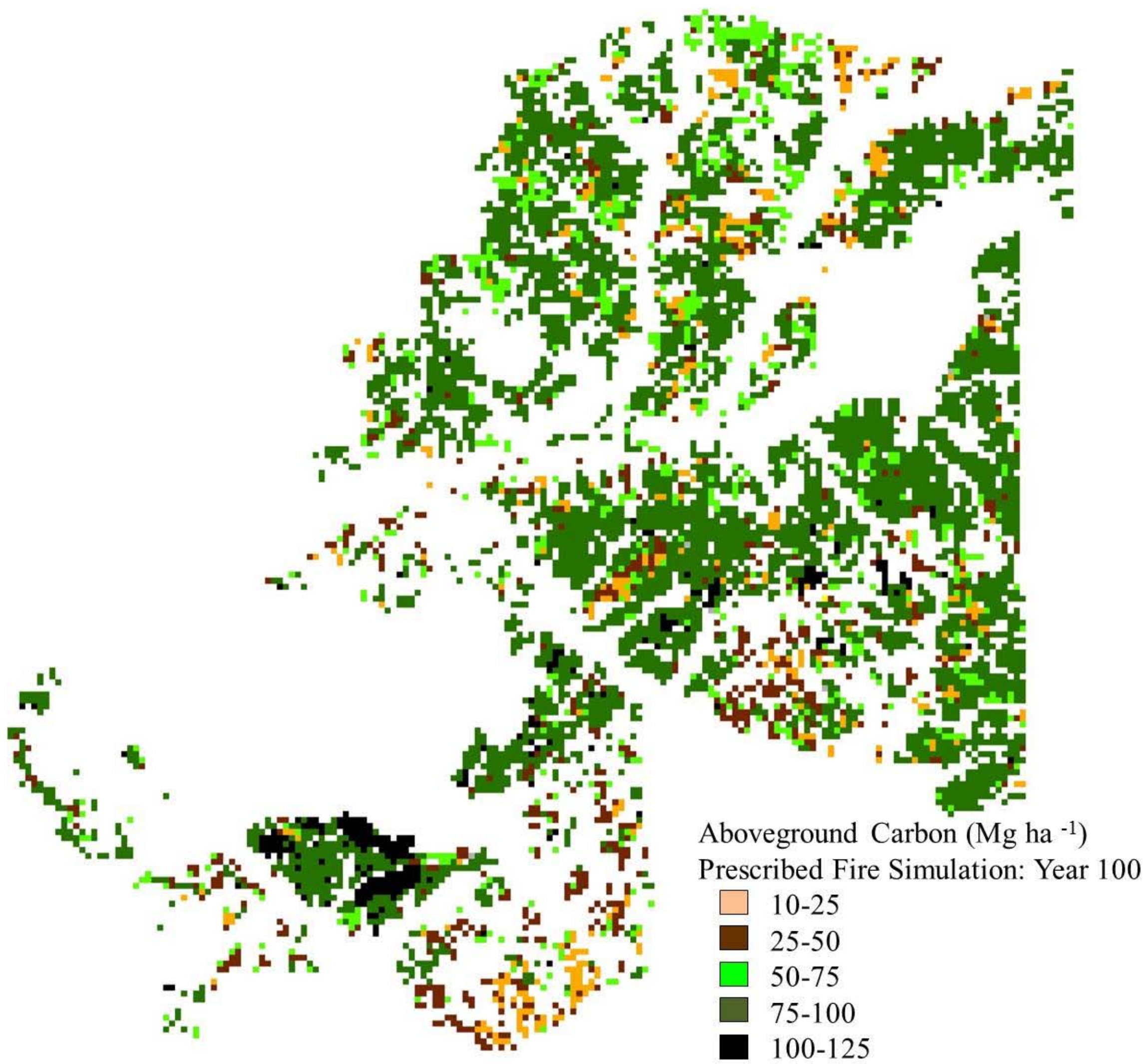

Figure 40: Year 100 aboveground carbon stock values simulated in LANDIS-II with the Century Succession extension at Fort Benning for the prescribed fire simulation.

Installation wide, total ecosystem carbon increased under all treatment scenarios (Figure 41). The control TEC behaved as expected, becoming asymptotic by the end of the simulation period. A pattern that has been found in numerous forest conditions when disturbance is absent. The burn-only treatment also became asymptotic and had a mean TEC value that was 12% lower than the control at the end of the simulation period. The thin and burn treatment had additional carbon reductions due to thinning treatments. The higher sustained NECB for the thin and burn (Figure 42) was realized in TEC by continued increases throughout the simulation period. The thin and burn mean TEC was 22% lower than the control at the end of the simulation period.

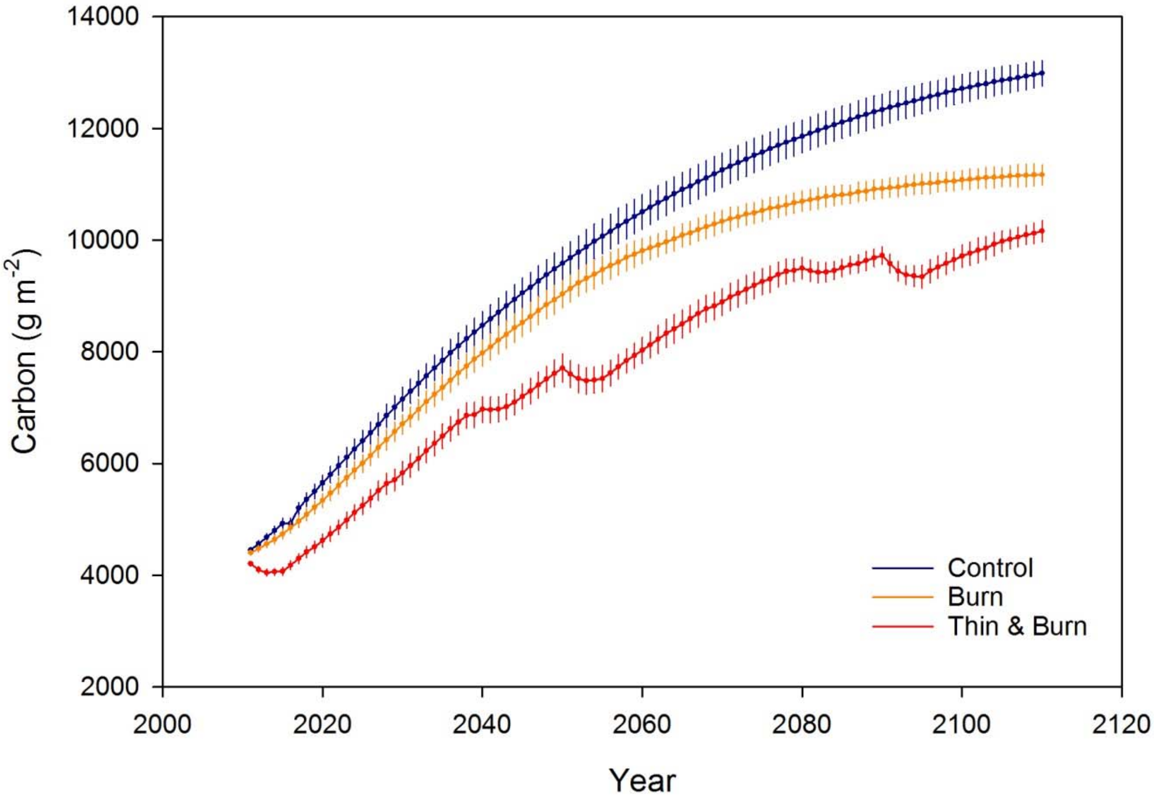

Figure 41: Total Ecosystem Carbon over 100 years for three treatment simulations. Values are mean and standard deviation from 100 simulation replicates.

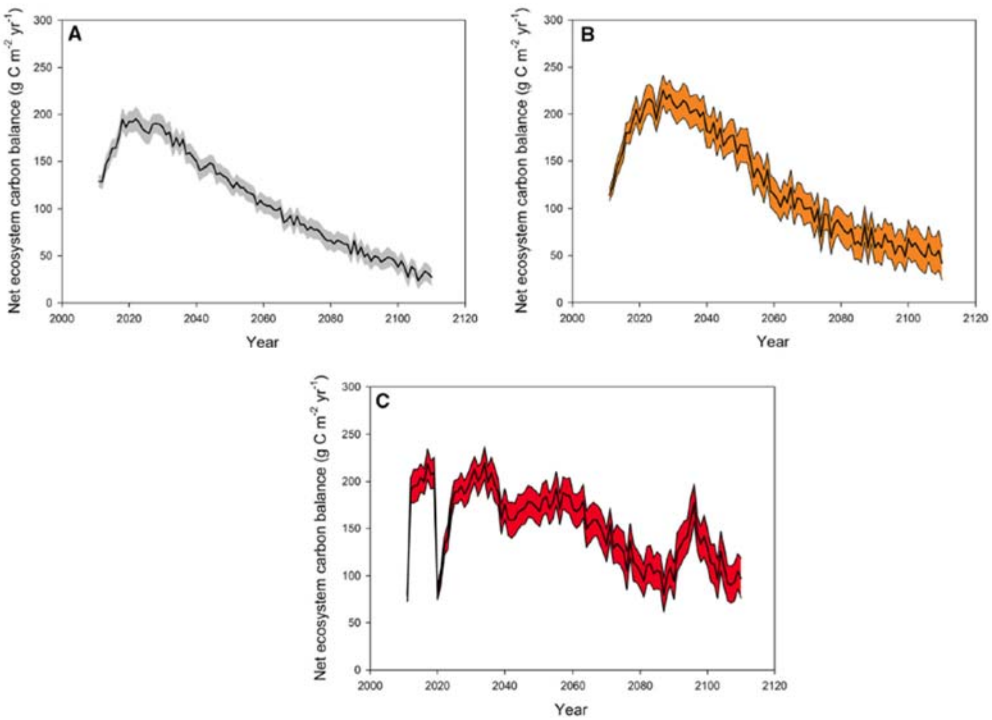

Figure 42: Net ecosystem carbon balance for the control (A), burn-only (B), and thin and burn (C) over the simulation period. NECB is calculated as net primary productivity, minus losses from heterotrophic respiration and disturbance. Values are means and 95% confidence intervals.

To test the effectiveness of treatments on reducing wildfire risk and the resultant effects on carbon dynamics, we ran a series of simulations over a 25 year simulation period. We shortened the simulation period because there is some evidence to suggest that a significant increase in broadleaved species can alter microclimatic conditions such that the ecosystem is less likely to burn (Nowacki and Abrams 2008). We parameterized the Dynamic Fire extension in LANDIS-II using historical ignition data for Fort Benning presented in Addington et al. (in press). We found that in the absence of longleaf pine restoration treatments and an active prescribed burning program, total ecosystem carbon declined substantially relative to the thinning and burning simulation (Figure 43). Total ecosystem carbon also declined in the thin and burn treatment, driven largely by the young age of restoration sites and the timing of wildfire events. There are two caveats that need to be considered when evaluating these results. First, we were unable to account for fire suppression efforts in these simulations. When an ignition occurs, the resultant fire size is a function of weather and fuels. Second, the short simulation period (25 years) does not provide information on forest recovery following large fire years. As we demonstrated in Martin et al. (2015), longleaf pine restoration treatments coupled with prescribed burning push the forest toward a more fire-adapted condition, albeit with a lower TEC state (Figure 41). However, the trajectories of the simulations with wildfire suggest that the restoration scenario is more high-severity fire resistant and could surpass the control TEC as the forests move toward more of the carbon being aggregated in fewer, larger, more fire-resistant trees (Figure 43).

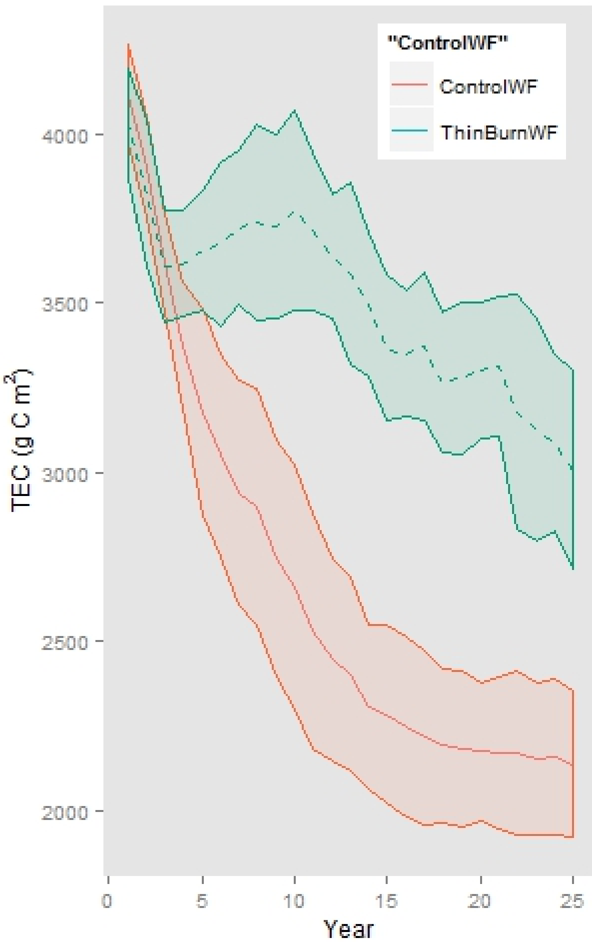

Figure 43: Total ecosystem carbon over a 25 year simulation period inclusive of stochastic wildfire for the control and thin and burn treatments. Values are means and 95% confidence bands.

LANDIS-II Simulations – Camp Navajo

To implement LANDIS-II simulations, CN was gridded using a 150 m grid divided into six ecoregions as a function of soil type and topography, because topography is a major determinant of climate in this region. We developed the initial forest communities layer using field inventory data collected as part of this research and age-size distributions from Fule et al. (1997) and Mast et al. (1999). We parameterized two species, *Pinus ponderosa* and *Quercus gambelii*, which accounted for greater than 99% of the biomass in our inventory plots. Species-specific parameters were obtained from the literature (Appendix I). Following model spin-up, the Camp Navajo landscape was a C sink with mean net ecosystem productivity (NEP) of 175 g C m^{-2} yr^{-1} (sd=53). Dore et al. (2012), using the eddy covariance approach, reported a range of NEP from 19 (76) g C m^{-2} yr^{-1} to 174 (57) g C m^{-2} yr^{-1} over a five-year period, inclusive of a year with significant drought. We used our inventory data and allometric equations from Jenkins et al. (2003) to calculate individual tree biomass and then scaled these values to a per unit area basis for comparison with simulated aboveground biomass values. Our inventory aboveground biomass ranged from 1917 to 25,645 g m^{-2}, with a mean value of 12,106 g m^{-2}. Our simulated aboveground biomass values ranged from 2084 to 14,032 g m^{-2}, with a mean value of 11,540 g m^{-2}.

We simulated common forest treatment practices in southwestern ponderosa pine, including understory thinning to reduce fuel continuity between the forest floor and canopy and prescribed burning to reduce surface fuel loads. We used the Leaf Biomass Harvest extension to implement both thinning and prescribed burning treatments following Syphard et al. (2011). We used this approach for prescribed burning to facilitate wildfire simulations using the Dynamic Fire extension because both prescribed fire and wildfire cannot be simulated simultaneously in the Dynamic Fire extension. We simulated understory thinning by preferentially targeting the youngest cohorts of trees following common forest restoration practice based on historical forest reconstructions (Fulé et al. 1997, Finkral and Evans 2008). We excluded from treatment areas with slopes > 14% as these areas are operationally difficult to treat and are often nest sites for the Mexican spotted owl (*Strix occidentalis lucida*) (Prather et al. 2008). We ran a series of wildfire simulations prior to implementing thinning treatments to identify geographic locations with the highest fire risk (Figure 44). We then ranked treatment implementation timing as a function of fire risk (Figure 45). We implemented thinning treatments on 12% of the installation per year, minus the excluded areas, until all areas identified for treatment were completed. To simulate prescribed burning with the Leaf Biomass Harvest extension, we implemented a treatment that removed 90% of 1-10 year old cohorts and a small fraction of older cohorts to simulate fire-induced mortality. Following the thin and burn, fuels were reduced and crown base height was increased to simulate consumption by fire. The prescribed fire treatment used a ten-year return interval.

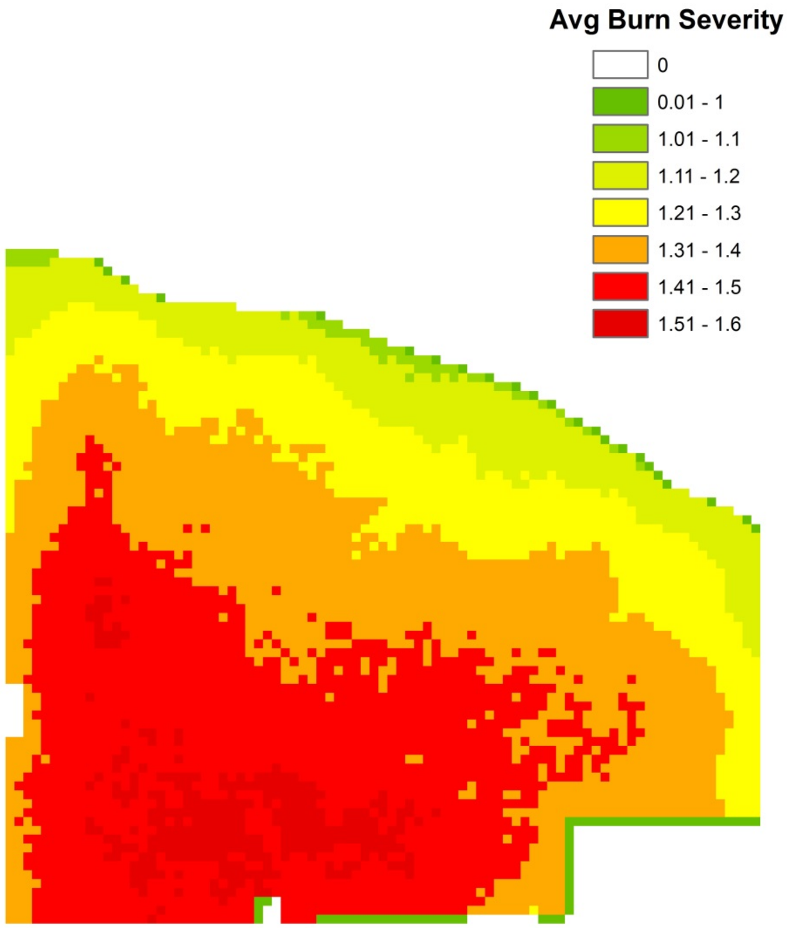

Figure 44: Mean fire severity from simulations with random ignition at Camp Navajo.

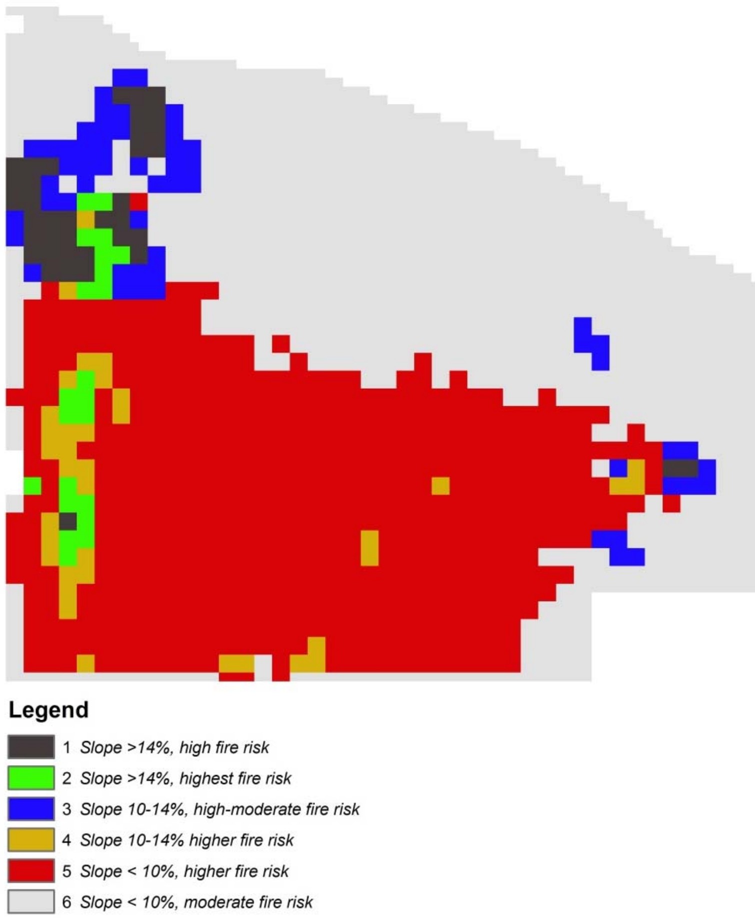

Figure 45: The Camp Navajo landscape categorized as a function of fire risk and slope. Areas with slopes >14% were excluded from treatment.

We used the Dynamic Fire and Fuels extension to simulate stochastic wildfire events across the installation. We used the Coconino National Forest wildfire database to obtain data to parameterize the fire size distribution, ignition frequency, and seasonality. Following Scheller et al. (2011b) we adjusted parameter values from the Canadian Forest Fire Behavior Prediction System using spread rates in Scott and Burgan (2005). We ran simulations with two different fire occurrence probabilities (1 in 50 chance and 1 in 100 chance of wildfire occurence). These probabilities represent the lower end of the historic range of regional large wildfire probability estimated by Dickson et al. (2006). We used data from the KFAST station for Flagstaff, AZ to provide fire weather data and Fire Family Plus (Bradshaw and McCormick 2000) to evaluate seasonality and severity as a function of weather conditions. We used this extension to produce spatial fire severity outputs for each time-step. Fire severity is categorical and ranges from one to five, with one being low severity surface fire and five being high severity. At a severity of three, fires begin to torch (burn up into tree canopies) and can initiate a crown fire.

To evaluate the effects of forest treatments on C dynamics we ran three different treatment scenarios; control, thin-only, thin and burn. Both the thin-only and thin and burn used an understory thin to remove approximately 30% of the live tree C, an approach common for this forest type (Hurteau et al. 2011). The thin and burn treatment included a simulated prescribed fire implemented with a ten year return interval, such that 10% of the installation was burned by prescribed fire every year. We ran each of these simulations with three levels of wildfire; no wildfire, ignition probability = 2% yr^{-1}, and ignition probability = 1% yr^{-1}. We ran 50 replicates of each scenario for 100 years to capture the stochastic nature of wildfire occurrence. To quantify the effects of forest treatments and wildfire on C stocks and fluxes, we calculated the mean and 95% confidence intervals for total ecosystem carbon (TEC) and net ecosystem carbon balance (NECB). NECB accounts for both carbon assimilation from net primary productivity and losses due to respiration and disturbance (Chapin and Matson 2011). Our NECB values do not include carbon removal from understory thinning in our simulations, because the fate of the thinned biomass is variable in this region (Finkral and Evans 2008). To determine the effectiveness of forest treatments on altering fire effects, we calculated the mean and coefficient of variation for fire severity outputs from the Dynamic Fire and Fuels extension for each scenario using all time-steps from the 50 replicates. Analyses of simulation data were conducted in R using the Raster package and figures were produced using the ggplot2 package (R Core Team 2012, Hijmans and van Etten 2012, Wickham 2009).

At Camp Navajo, TEC in the absence of wildfire was consistently higher in the control than in either of the treatments (Figure 46). We had expected a sustained reduction in TEC with both treatments and a larger reduction for the thin and burn treatment. However, thin-only and thin and burn TEC had relatively small differences over the majority of the 100 year simulation. Treatments reduced mean TEC by approximately 100 g C m^{-2} below the control in the absence of wildfire at the end of the 100-year simulation period (Figure 46). Thin-only and thin and burn treatments enhanced NECB relative to the control over the first half of the simulation period (Figure 47). The TEC and NECB results are likely due to growth release that occurs from thinning in southwestern ponderosa pine (Kerhoulas et al. 2013, McDowell et al. 2006). In simulations that included wildfire, the thin and burn treatment reduced both mean fire severity and its coefficient of variation (Figures 48 and 49). On the western edge of the landscape are areas excluded from thinning because of steep slopes and potential Mexican spotted owl habitat. The effect of slope interacting with fuels on fire severity is evident across all three scenarios, as these steeper areas had higher mean fire severity (Figure 48). However, mean severity tended to be lower than other treatments in the thin and burn because of the effect of this treatment on slowing fire spread. In the thin and burn, the areas excluded from treatment also had increased fire severity coefficient of variation relative the remainder of the landscape (Figure 49c). When the probability of fire occurrence was simulated at 2% yr^{-1}, the thin and burn treatment had substantially higher TEC than the control and thin-only by the end of the simulation period (Figure 50). When the probability of fire occurrence was simulated at 1% yr^{-1}, the control had higher TEC for the first half of the simulation period, but was surpassed by the thin and burn during the second half of the simulation period (Figure 50). When we included wildfire in the simulations, the thin-only and thin and burn had enhanced NECB, while the control NECB decreased more rapidly with wildfire than in the absence of wildfire (Figure 51).

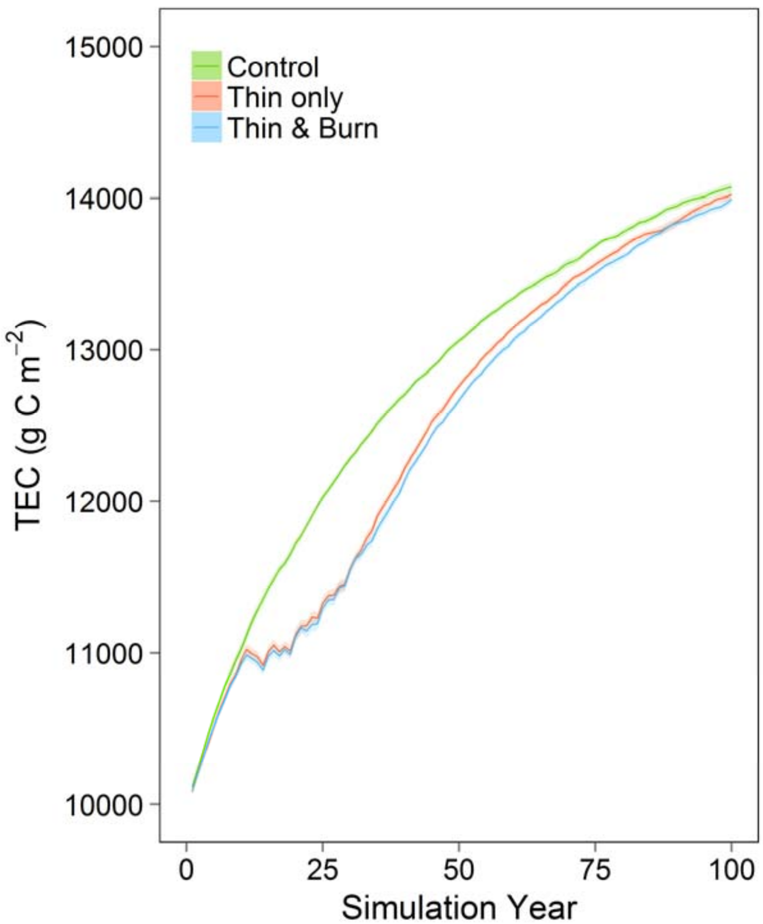

Figure 46: Total ecosystem carbon (TEC) for the three simulated treatments (control, thin-only, thin and burn) in the absence of simulated wildfire over the 100-year simulation period. The dark lines represent mean TEC by treatment for 50 simulation replicates.

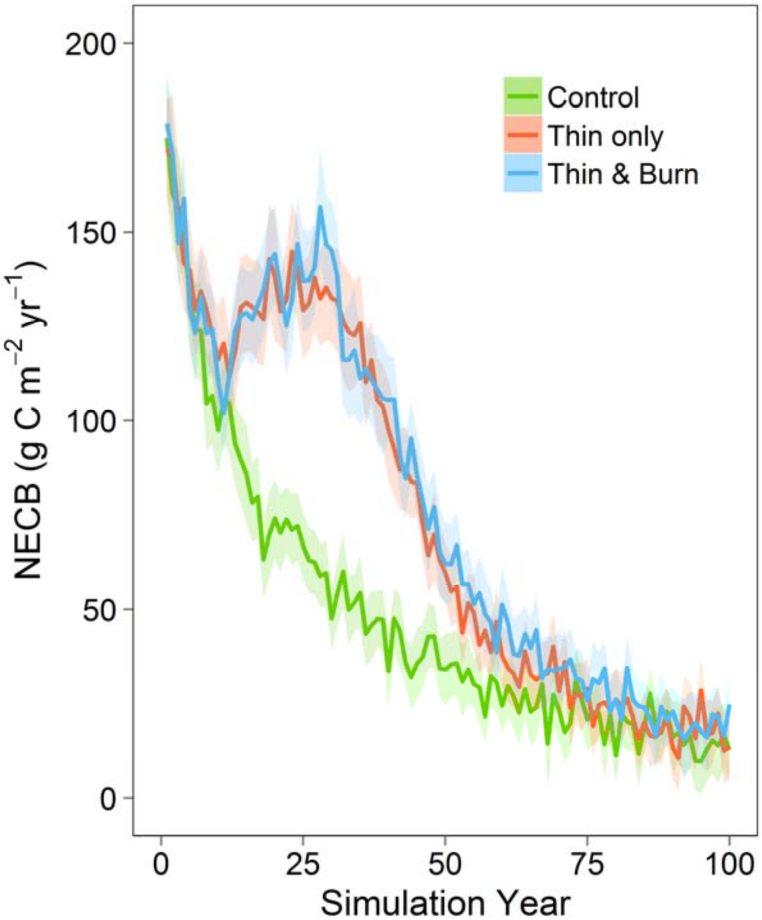

Figure 47: Net ecosystem carbon balance (NECB) for the three simulated treatments the absence of wildfire over the 100-year simulation period. The dark lines represent mean NECB and the shaded area 95% confidence intervals.

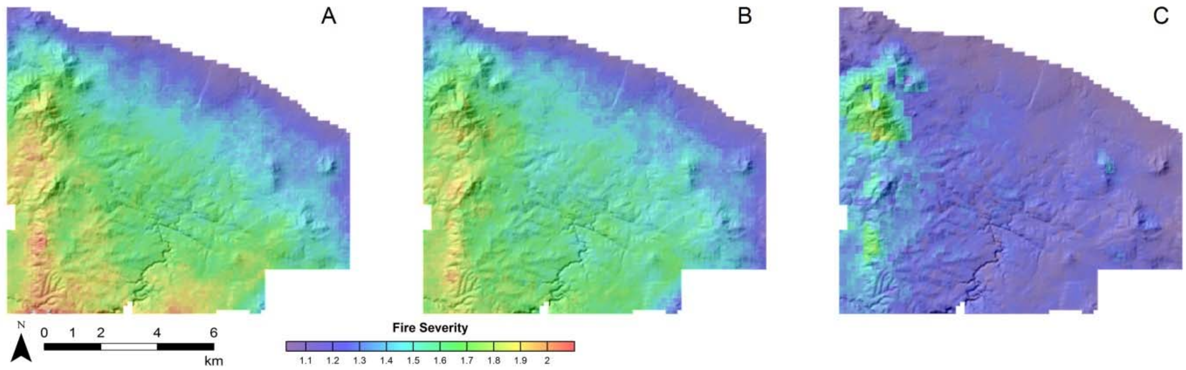

Figure 48: Mean fire severity calculated from the 50 simulation replicates at Camp Navajo, AZ for the control (A), thin-only (B), and thin and burn (C) using a probability of fire occurrence equivalent to 0.02.

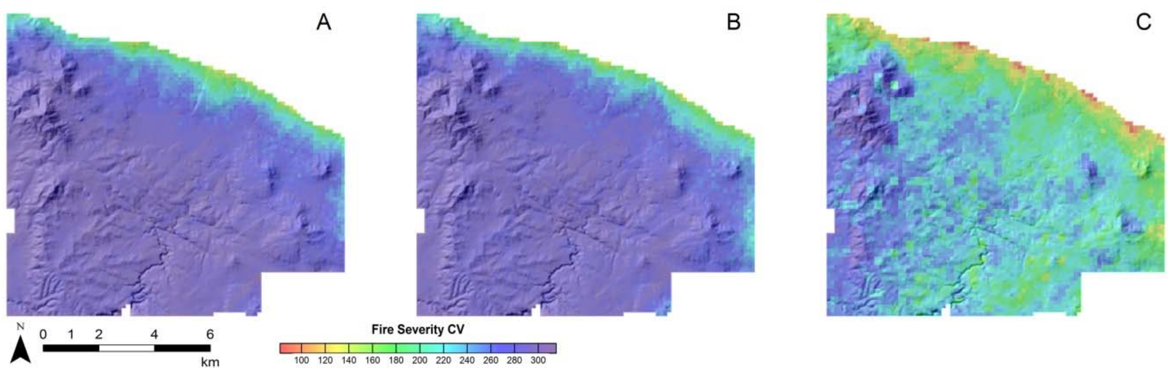

Figure 49: Coefficient of variation (CV) of fire severity across Camp Navajo, AZ for the control (A), thin-only (b), and thin and burn (C) using a probability of fire occurrence equivalent to 0.02.

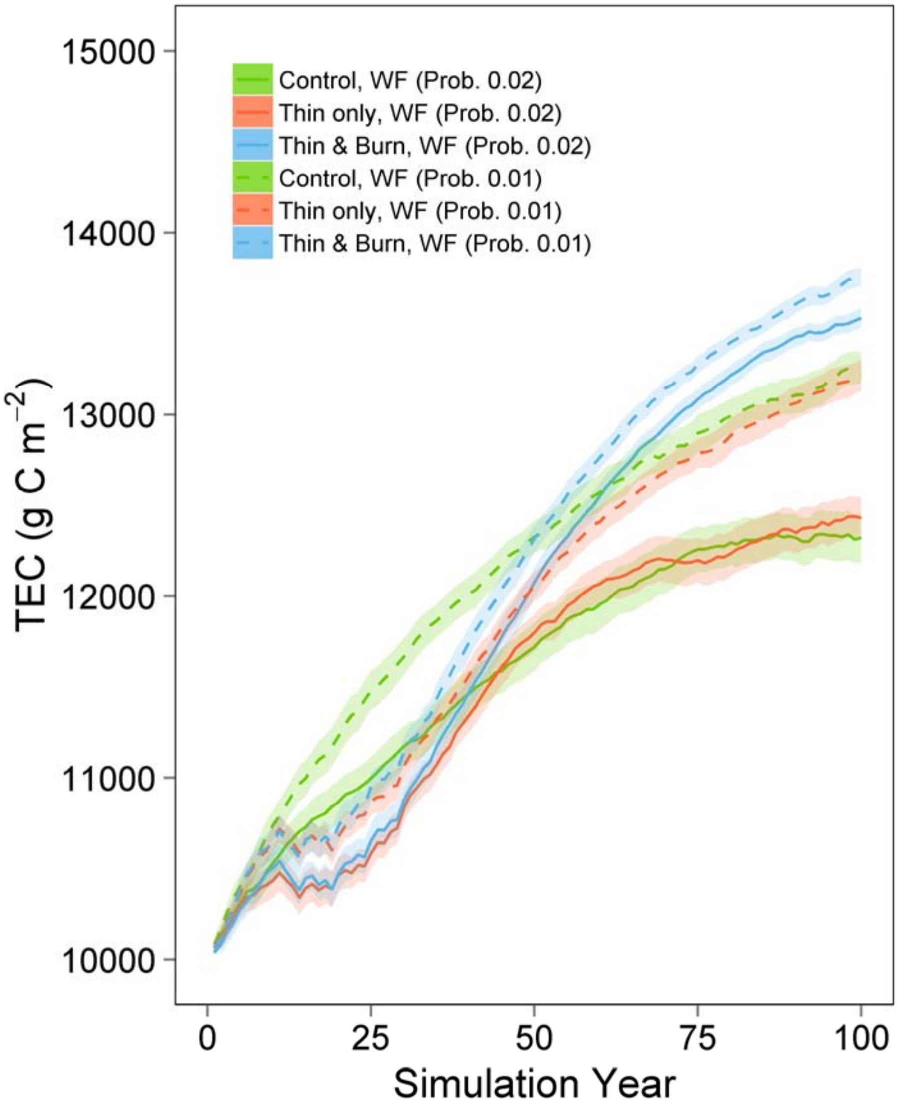

Figure 50: Total ecosystem carbon (TEC) for the three simulated treatments with the probability of wildfire occurrence simulated at 2% yr-1 and 1% yr-1 over the 100-year simulation period. The dark lines are mean TEC and shaded areas 95% confidence intervals.

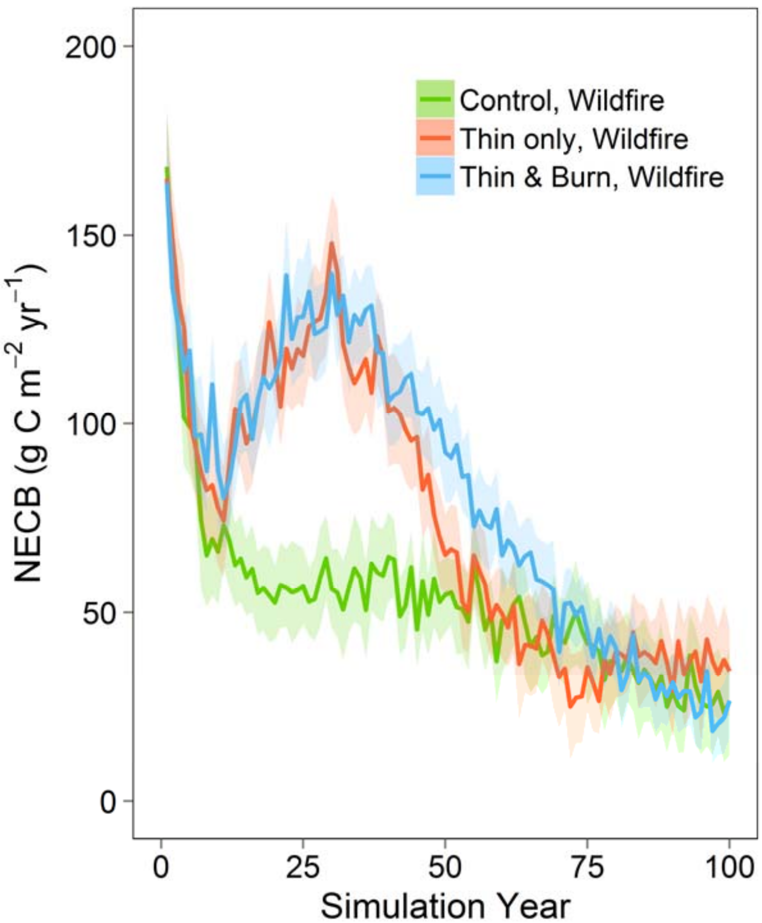

Figure 51: Net ecosystem carbon balance (NECB) for the three simulated treatments with the probability of wildfire occurrence simulated at 2% yr-1 over the 100-year simulation period. The dark lines are mean TEC and shaded areas are 95% confidence intervals.

LANDIS-II Simulations – Joint Base Lewis McChord

To implement LANDIS-II simulations, JBLM was gridded using a 200 m grid divided into two ecoregions as a function of soil type. We developed the initial forest communities layer using the Landscape Ecology, Modeling, Mapping & Analysis (LEMMA) Laboratory's Washington Coast and Cascades gradient nearest neighbor (GNN) interpolated map (221) and database (LEMMA 2006, Ohmann and Gregory, 2002). The GNN technique incorporates regional grids of Forest Inventory Analysis (FIA) and USFS Current Vegetation Survey (CVS) plot data, spatially-explicit environmental data (such as geology, topography, climate), and Landsat Thematic Mapper (TM) satellite imagery to predict forest composition at the landscape scale (Ohmann and Gregory 2002). The LEMMA GNN product has 30 m resolution. We used Arcmap 10.1 (ESRI 2012) to resample the GNN-based initial communities layer to decrease pixel resolution from 30 m^2 to 200 m^2 using a nearest neighbor resampling technique, and further reduced the number of unique communities by binning similar species composition and cohort ages. The aggregated layer included all listed GNN map species that occurred in at least 1% of the grid cells and occupied at least 10% of the basal area in a given GNN grid cell. The parameterized species include: *Abies grandis, Acer macrophyllum, Alnus rubra, Fraxis latifolia, Pinus ponderosa, Pseudotsuga menziesii, Quercus garrayana, Thuja plicata,* and *Tsuga heterophylla*. Species-specific parameters were obtained from the literature (Appendix I). We used our field data and genus-specific allometric equations (Jenkins et al. 2003) to estimate aboveground biomass for comparison with simulation data. We subtracted 100 years from current cohort ages and ran simulations from 1912 to 2012. From this output, we extracted year 2012 aboveground biomass (AGB) values from the 5533 pixels that contained forests. Plot data biomass ranged from 1510 to 82,414 g m^{-2}, with a mean of 36,988 g m^{-2}. Simulated biomass ranged from 416 to 65,535 g m^{-2}, with a mean of 41,100 g m^{-2}.

We used a full factorial design to quantify the effects of management actions on carbon dynamics. We simulated four management scenarios: control (no management), thin only, burn only, and thin and burn. Thinning treatments were designed to remove 40% of the net annual growth (45,000-54,000 m^3 biomass yr^{-1}, Griffin 2007) and maintain forest cover. We used the Leaf Biomass Harvest extension to implement both thinning and prescribed burning treatments following Syphard et al. (2011). We used this approach for prescribed burning to facilitate wildfire simulations using the Dynamic Fire extension because both prescribed fire and wildfire cannot be simulated simultaneously in the Dynamic Fire extension. The thinning treatment targeted 70-180 year old cohorts of Douglas-fir and was simulated to occur on 1% of the land area with prairie soils and 0.5% of the area with non-prairie soils annually. We included 2% removal of cohorts of other species to simulate mechanical damage during harvesting operations. Within any given stand, thinning was limited to one entry during the simulation period and only in areas that had not been burned within the past 10 years. To simulate prescribed burning with the Leaf Biomass Harvest extension, we implemented a treatment that removed 85% of 1-26 year old cohorts of Douglas-fir, 90% of 1-5 year old cohorts of Oregon white oak, and 90% of 1-

15 year old cohorts of all other species. We limited prescribed fire treatments to prairies and Douglas-fir forest occurring on prairie soils. We excluded areas with trees > 300 years old, stands with late-successional conifer species, and riparian buffer areas from prescribed burning treatments. Prescribed burning in prairies had a five-year return interval and Douglas-fir forest on prairie soils had a twenty-year return interval.

To examine the effects of intensive management activities on Oregon white oak restoration, we developed an oak restoration treatment. We identified a 708 ha area that had prairie soil, did not include riparian areas, and included extant oak in which to test the effects of intensive management on increasing the probability of establishing oak savannah (Figure 52). Within this intensive management area, we simulated thinning and burning to reduce conifer competition to a level that would allow oak regeneration. The thinning treatment removed 85% of non-pine conifers and included regular burning.

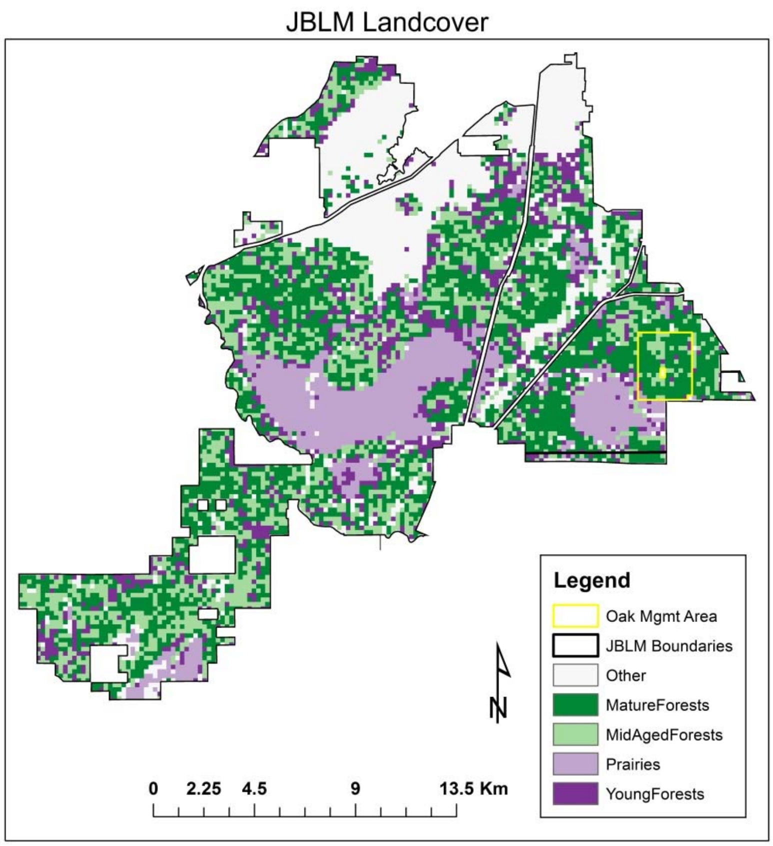

Figure 52: Map of broad forest type classification for Joint Base Lewis McChord. The map includes the location of the 708 ha intensive Oregon white oak management area.

We used the Dynamic Fire and Fuels extension to simulate stochastic wildfire events across the installation. We used wildfire occurrence data from JBLM prairie fires (2007-2011) and US Forest Service forest fires within 100 km of JBLM (1995-5005) to obtain data to parameterize the fire size distribution, ignition frequency, and seasonality. Following Scheller et al. (2011b) we adjusted parameter values from the Canadian Forest Fire Behavior Prediction System using spread rates in Scott and Burgan (2005).

To quantify the effects of treatment on carbon pools and fluxes, we calculated net ecosystem carbon balance (NECB) and total ecosystem carbon (TEC). To quantify the installation-level effects of intensive management to favor oaks in the 708 ha management unit, we ran the thin and burn scenario at the landscape-level, with intensive management within the oak management unit.

At JBLM, the relatively young age of most forested areas lead to NECB increasing for the first 20 simulation years in the absence of treatment (Figure 53). As the forest matured the sink strength declined because of increased respiration. When we simulated the thinning alone, NECB peaked higher than in the control and was sustained at a higher rate than the control over the entire length of the simulation period. This result is due to the release effect from thinning, whereby the remaining trees are released from competition for resources. The burn only and the thin and burn treatment had the effect of decreasing the sink strength of the forest relative to the control for the first 30 simulation years, at which point the thin and burn had a sink strength to the thin only and the burn only had a slightly higher sequestration rate than the control throughout the remainder of the simulation period. (Figure 53).

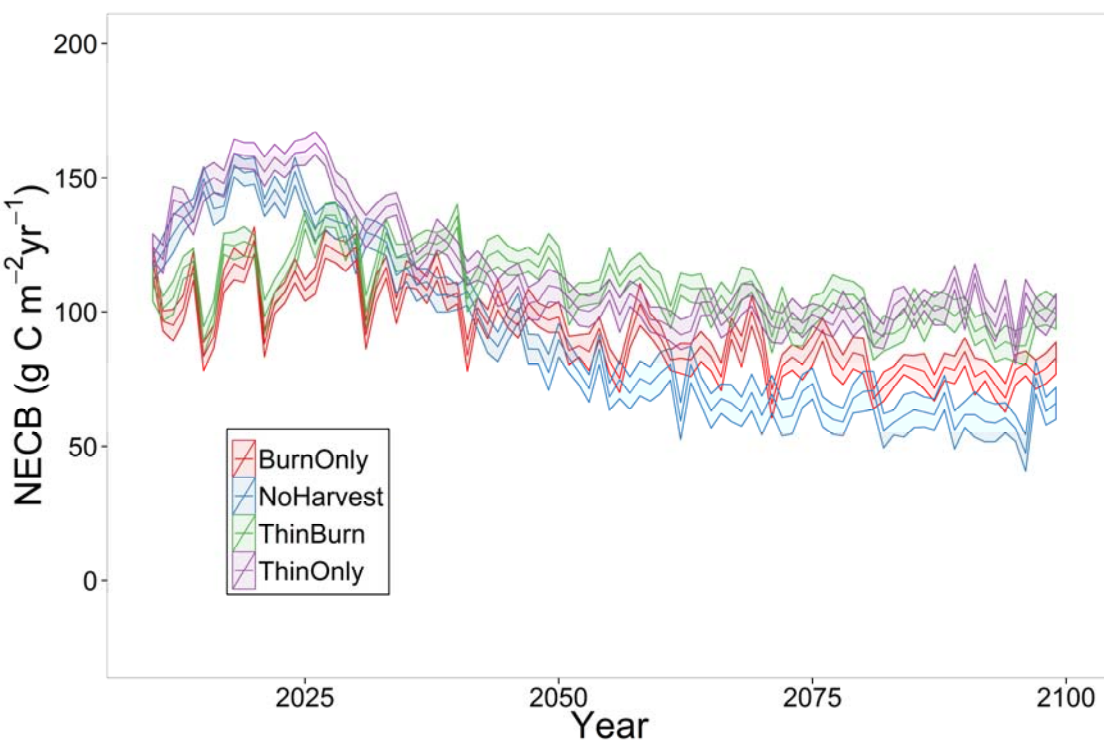

Figure 53: Net ecosystem carbon balance (NECB) for the four simulated treatments over the 100-year simulation period. The dark lines represent mean NECB and the shaded area 95% confidence intervals.

As expected, total ecosystem carbon (TEC) decreased with increasing management intensity (Figure 54). Under the control scenario, TEC increased by approximately 50 Mg C ha^{-1} over the simulation period. The burn-only and thin and burn increased at similar rates to the control, but with lower initial C stocks. The thin-only treatment, which was based on current sustainable harvesting practices, accumulated approximated 25 Mg C ha^{-1} over the simulation period and had a TEC 8.8% below the control by the end of the simulation period (Figure 54). All simulations included the probability of wildfire occurrence developed from the JBLM prairie fire data and USFS forest fire data. The low probability of wildfire occurrence within the forested areas, coupled with high canopy base height for these Douglas-fir dominated forests resulted in wildfire having insignificant impacts on forest carbon dynamics.

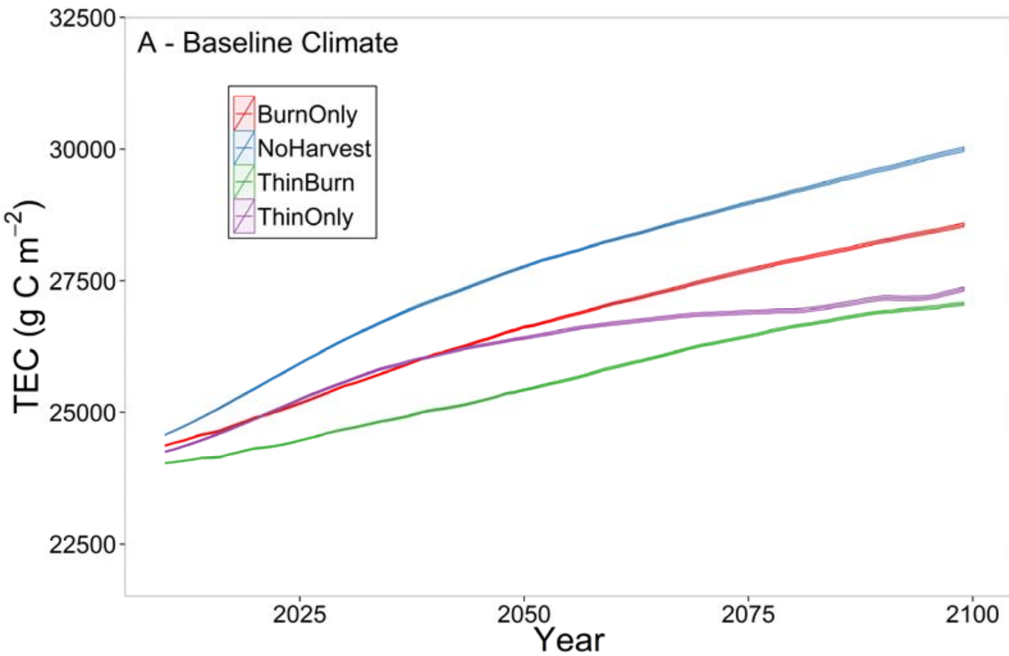

Figure 54: Total ecosystem carbon (TEC) for the four simulated treatments over the 100-year simulation period. The dark lines represent mean NECB and the shaded area 95% confidence intervals.

To determine the effects of treatment on the probability of Oregon white oak occurrence at the end of the simulation period, we constructed probability surfaces using the 50 replicate simulations for each treatment scenario. We compared the empirical distribution functions for each treatment against the control using the Kolmogorov-Smirnov test. We found that the thin-only and thin and burn treatments both increased the probability of oak occurrence ($p<0.0001$) relative to the control (Figure 55).

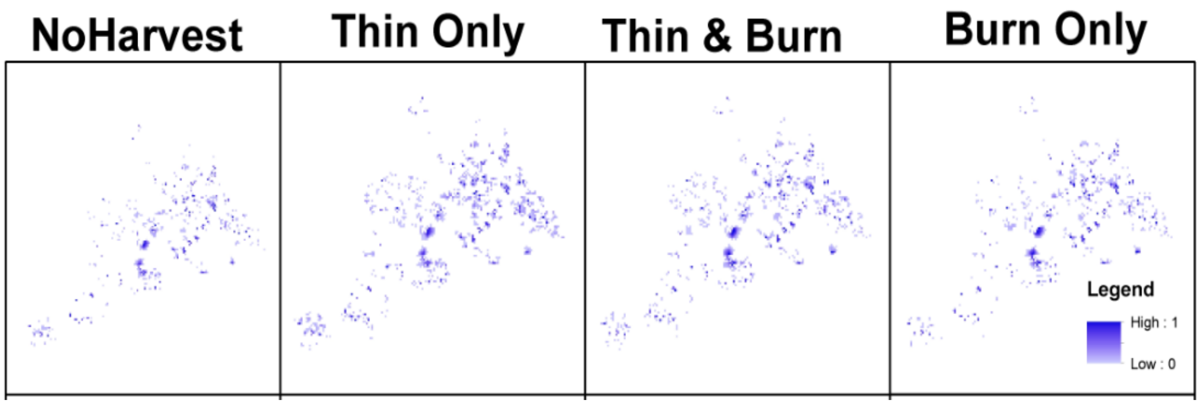

Figure 55: Probability of Oregon white oak occurrence at the end of the 100 year simulation period under four different treatment scenarios.

Given the importance of the forest community type associated with the Oregon white oak, we investigated the potential for intensive targeted management. Within the 708 ha intensive management area, we found that through increased harvesting intensity focused on Douglas-fir, coupled with regular burning, the probability of oak occurrence increased substantially (Figure 56). Given the size of the intensive treatment area relative to the total installation size, the impact of the intensive treatment on total ecosystem carbon was negligible relative to the thin and burn treatment.

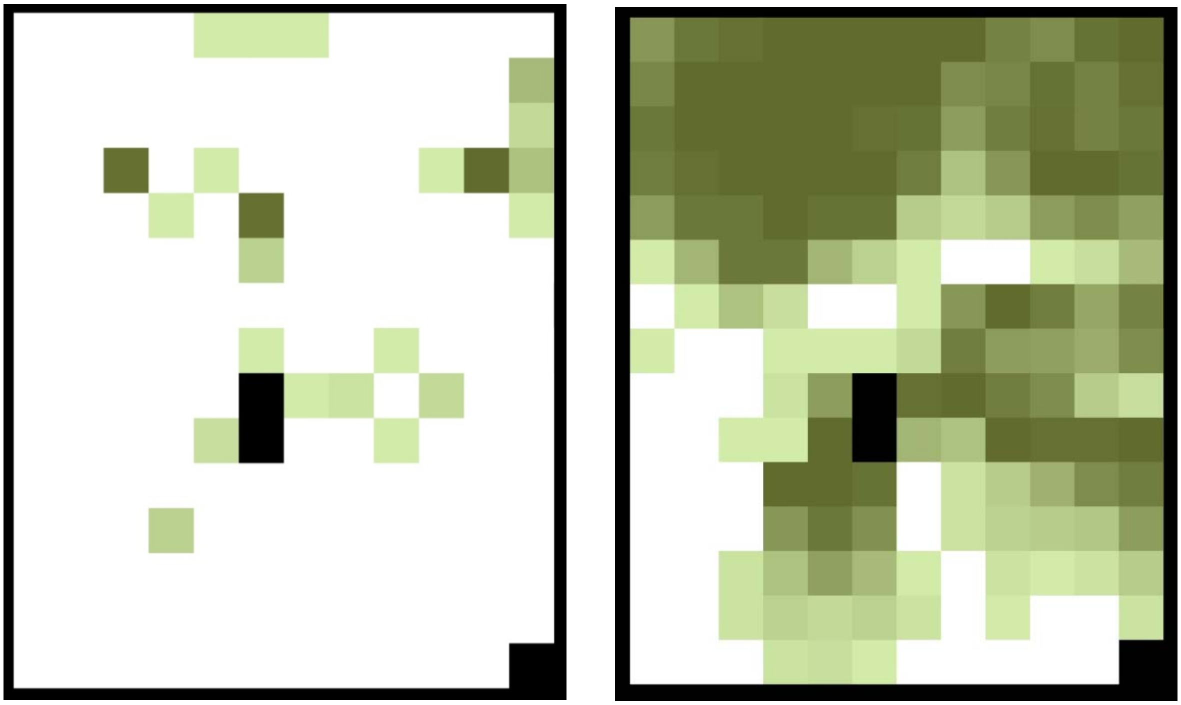

Figure 56: Probability surfaces for the 708 ha oak management area. The figure on the left is the thin and burn. The figure on the right is the intensive oak management treatment. The color scale ranges from 0 (white) to 1 (dark green).

Wildlife Results – Fort Benning

At Fort Benning, plots were widely distributed across the installation in an effort to sample a range of RCW habitat (Figure 57). We examined stand attributes associated with RCW use and were surprised that there was not a close associate with stand age or basal area (Figure 58). There were, however, three stand variables that were associated with RCW use: herbaceous cover, woody debris cover, and number of pine snags (Figure 59).

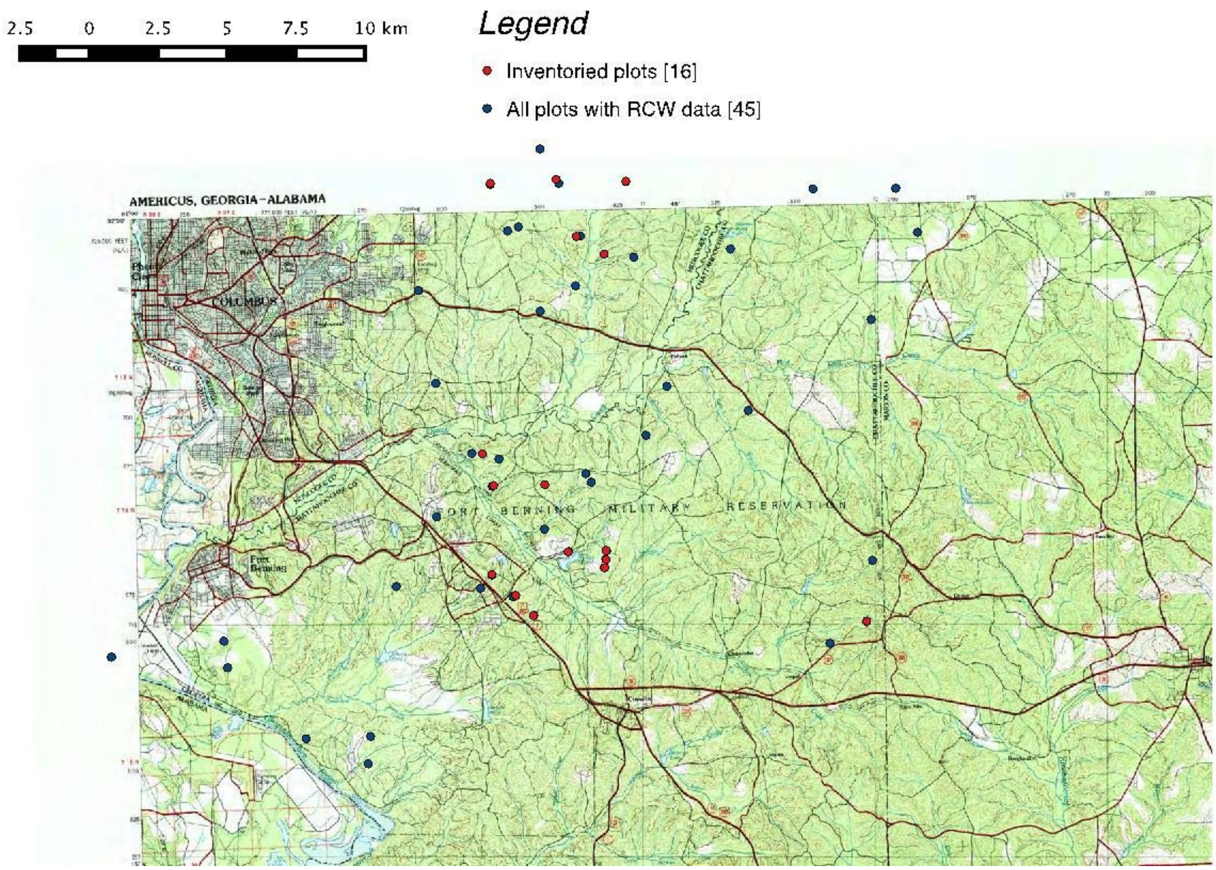

Figure 57: Red-cockaded woodpecker plot distribution at Fort Benning.

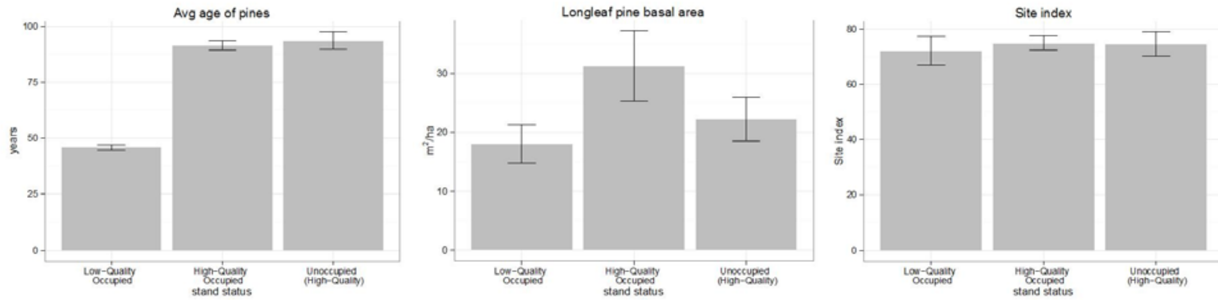

Figure 58: Mean stand age (left), longleaf pine basal area (center), and site index (left) for the three red-cockaded woodpecker habitat types sampled at Fort Benning.

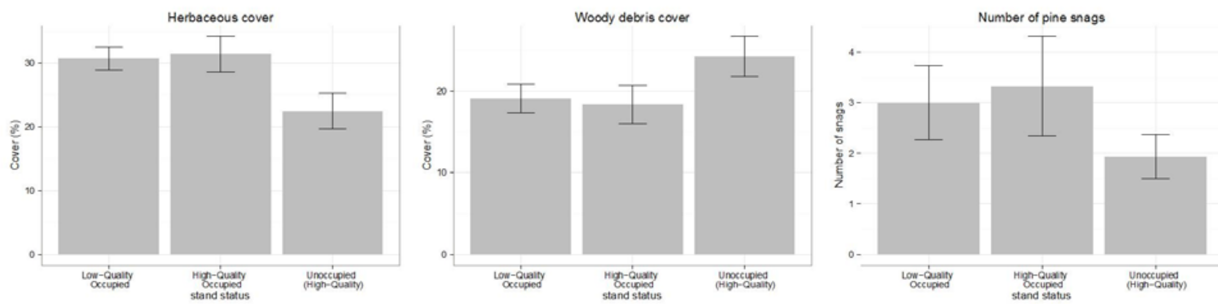

Figure 59: The three stand attributes associated with red-cockaded woodpecker use were herbaceous cover (left), woody debris cover (middle), and number of pine snags (right).

The RCW appears to respond to understory cover and snag density. We did not find statistically different values for these variables between "high-quality" and "low-quality" occupied stands. We also evaluated spatial structure of nest tree stands. Trees were mapped within each sampling plot (Figure 60).

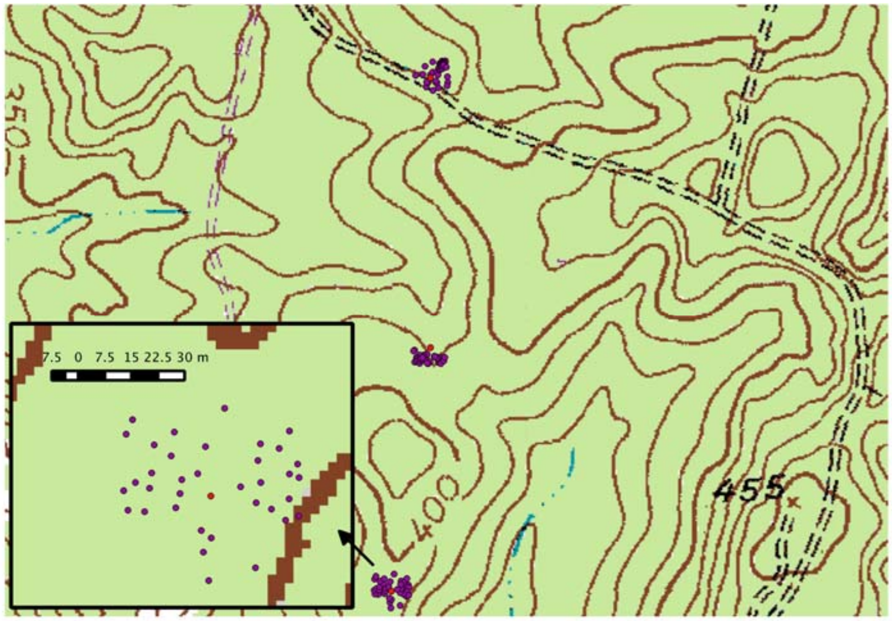

Figure 60: Example of mapped tree locations (purple dots) around a known RCW next tree (red dot) at Fort Benning.

Wildlife Results – Camp Navajo

At Camp Navajo, analysis of tree radial growth response to different treatments and residual clump sizes and densities suggests there is not a significant reduction in growth potential. We did not find a consistent change in pre- and post-thinning mean annual BAI that was correlated with the density level around the 'target' tree (Figure 61). This result was unexpected because higher local density is usually associated with increased competition for growth resources (particularly water in ponderosa pine) and reduced growth. We did not make physiological measurements and therefore can only speculate about the putative causes for this result. All measured clumps bordered gap areas created by removing all trees. It may be that competition for soil moisture is reduced because tree roots extend into adjacent gaps to acquire more moisture. This may mean that adjacent gap creation is an important driver of clump tree growth response.

Our results suggest that thinning treatments do accelerate the radial growth rate of leave trees (>75% of post-treatment growth values are >1 in Figure 61) and therefore the development of large trees associated with preferred Mexican spotted owl habitat. Our results also show there was no significant reduction in radial growth increment associated with the stem density immediately around the measured 'target' tree. This suggests that retaining tree clumps designed to provide high canopy cover associated with owls, while also accelerating large tree development, may be possible if thinning produces gaps or low tree-density areas immediately surrounding the clumps of potential future owl habitat.

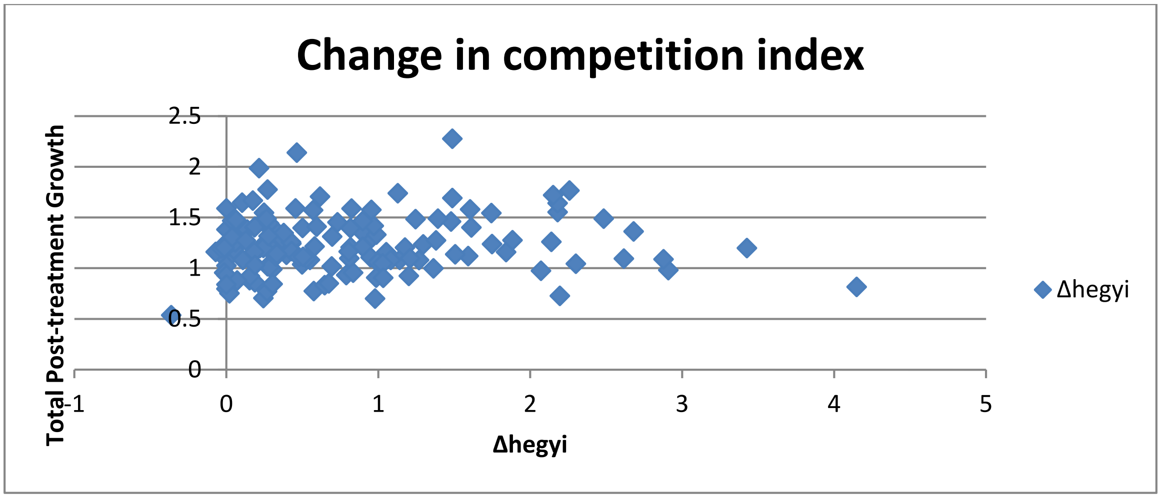

Figure 61: The ratio of post- and pre-treatment growth (average 5-year post BAI/average 5-year pre BAI) plotted against the change in relative stand density (as measured by the Hegyi index). Higher delta Hegyi values indicate a greater reduction in density.

Wildlife Results – Joint Base Lewis McChord

At JBLM preferred gray squirrel stands have high levels of crown-to-crown connectivity and a lower percentage of the plot area in gaps. Interestingly canopy cover and quadratic mean diameter (QMD) did not differ significantly between preferred and avoided habitat (Table 7). Past thinning has moderately decreased canopy cover and increased tree size (by thinning from below and removing smaller trees). Removing these trees and leaving overstory trees that are more regularly spaced, however, has increased gap areas and reduced canopy continuity.

Table 7: Comparison of mean structural attributes between stands with and without western gray squirrel use. * indicate values between the two use categories that are significantly different ($p<0.05$ t-test).

Gray Squirrel Use:	# of continuous crown paths	% plot area in gaps	Canopy cover	QMD (in)
Preferred	12.4*	8.4*	82	16.4
Avoided	3.6	26.2	71	20.8

We used stand structural data and stem maps to build models of canopy connectivity in ArcGIS by first plotting the tree locations (Figure 62 left), using allometric equations to 'build' tree crowns (Figure 62 middle), and then joining the adjacent crowns (Figure 62 right).

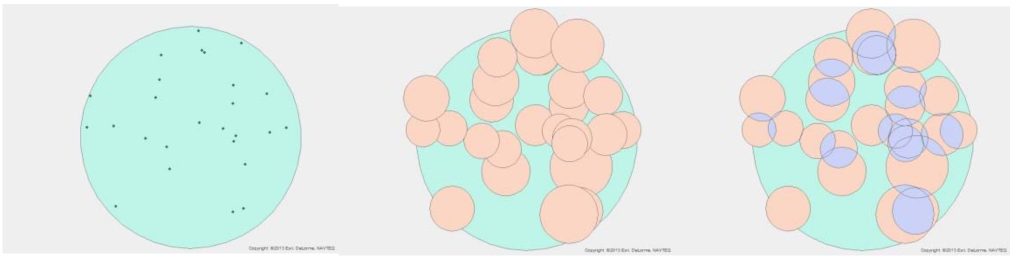

Figure 62: Process of canopy connectivity modeling. Stem mapped trees are plotted (left) and then allometric relationships are used to build tree crowns (middle). Connected crowns are then joined (right, purpled shaded areas) to identify connected canopies.

We used canopy connectivity models to calculate the number of exits out of each tree group that would allow western gray squirrels to travel out of the plot area. For the exit pathway analysis, canopy overlaps (Figure 62 right, purple shading) are dissolved to form tree group outlines (Figure 63).

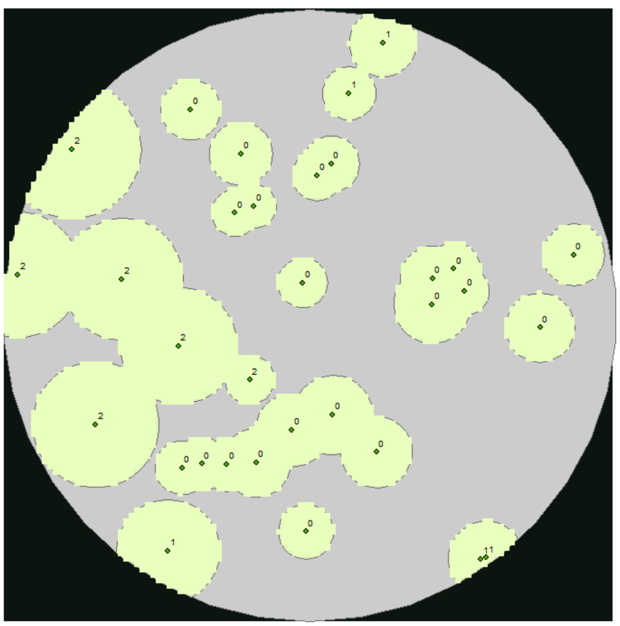

Figure 63: Example of dissolved canopies for exit pathway analysis used to determine the number of exits available for the western gray squirrel to leave a plot.

We found a difference between plot types with preferred squirrel stands having higher levels of crown-to-crown connectivity and lower percentages of plot areas in groups. Interestingly, canopy cover and quadratic mean diameter were not significantly different (Table 7). Past thinnings have moderately decreased canopy cover and increased tree size by thinning from below to remove smaller trees. Removing these trees and leaving overstory trees that are more regularly spaced has increased gap areas and reduced canopy continuity.

Conclusions and Implications for Future Research/Implementation

The results of our research suggest that C tradeoffs are associated with various management objectives. Taking no action resulted in the largest aboveground C stock across installations. However, maximizing the aboveground C stocks carries the costs of reduced wildlife habitat quality at Fort Benning and Joint Base Lewis McChord and high risk of stand-replacing wildfire at Camp Navajo. In general, treatments that included both thinning and burning resulted in the lowest aboveground C stocks over the simulation period. However, the combined thinning and burning treatments resulted in stand development that increased focal species habitat and reduced wildfire risk. At Fort Benning and Camp Navajo, thinning treatments were a one-time cost in terms of C stock reductions, whereas at Joint Base Lewis McChord a sustainable harvest of 40% of net annual growth resulted in a consistent C stock cost of harvest throughout the simulation. It is important to note that these carbon stock outcomes do not include carbon storage in long-lived wood products, which can store C for substantial time periods. Total ecosystem carbon results from this study represent the lower bound of carbon storage because we do not account for any wood product storage. Estimating C storage in wood products requires knowledge of local wood products markets, which is beyond the scope of this project.

The other important consideration for carbon dynamics and forest management treatments is the effects of management on carbon flux. We evaluated carbon flux by calculating net ecosystem carbon balance (NECB), which is net primary productivity minus heterotrophic respiration and emissions from disturbance. Our results demonstrate that across all three installations, treatments result in an increase in the strength of the carbon sink. At Fort Benning, burning alone resulted in higher NECB than the control. However, similar to the control the burning treatment NECB decreased over time. The thin and burn NECB fluctuated as a function of the timing of thinning, but was sustained at a higher rate than both the control and burning treatments over the entire simulation period. At Camp Navajo, both the thin-only and thin and burn treatments increased NECB over the control for the first half of the simulation period and NECB for all three treatments converged in the second half of simulation period as the effects of competitive release from thinning diminished. This was largely due to the release from competition that occurs with thinning and that the thinning treatments were all implemented in the first 12 years of the simulation. At Joint Base Lewis McChord the thin-only treatment pushed NECB above the control, whereas the burn-only and thin and burn caused an initial decrease because of the additional losses from prescribed burning. Because NECB peaks by year 20 of the simulation in the control and begins to decline due to forest aging, the thin-only and thin and burn were stronger sinks than the control throughout the majority of the simulation period.

Fire emissions from repeated prescribed burning are a source of emissions from maintaining forest structure at each of these installations. An important finding from this research is that these regular emissions from burning are small and resequestered by tree growth and regrowth in the understory between fire events. Furthermore, these regular emissions need to be considered in the context of the emissions and lost net primary productivity associated with high-severity wildfire. Previous work has found that wildfire emissions and fire-induced tree mortality can have substantial impacts in terms of both direct (combustion) and indirect (decomposition) C emissions (Dore et al., 2008; Wiedinmyer and Hurteau, 2010; Hurteau, 2013) and treatments to reduce high-severity wildfire risk yield reduced tree mortality and direct emissions (Hurteau and North, 2009; Stephens et al., 2009; North and Hurteau, 2011). Camp Navajo, the installation with the highest risk of stand-replacing fire and the least productive of the three installations, had a sustained sink with prescribed burning. At Fort Benning, we found that including wildfire pushed total ecosystem carbon in the control below that of the longleaf pine restoration simulation that included regular prescribed burning. Wildfire probability at Joint Base Lewis McChord is quite low. As a result, prescribed burning has less of an impact on fire risk and is a more relevant management tool in the context of meeting specific structural and composition objectives.

The results from our investigation into species-specific forest structural attributes that increase patch use suggest that there may be important attributes that are less commonly considered. At Fort Benning, our results suggest that herbaceous cover and snag density are important attributes for RCW habitat. Herbaceous cover typically correlates with canopy cover and we found that occupied stands, whether “high” or “low” quality, had more widely spaced overstory trees than unoccupied stands. At Joint Base Lewis McChord, canopy connectivity appears to be an important factor for patch use by the western gray squirrel. At Camp Navajo, dense patches typically left during thinning to provide structural heterogeneity and potential nesting habitat for the Mexican spotted owl did not differ in post-treatment tree growth from trees that were not left in clumps. Results at all three installations require additional investigation, but suggest that there are some forest attributes that may serve as useful metrics for assessing habitat quality.

The results of this research should be considered in the context of the uncertainty associated with future changes in climate and its effects on forest growth and wildfire. Changing climate has the potential to influence post-fire C dynamics through reduced growth or vegetation type conversion (Hurteau and Brooks, 2011). This potential could change the risk evaluation equation by increasing both the probability of wildfire occurring and the consequence of wildfire on forest C dynamics. Military installations, in addition to having natural ignition sources, also have an increased probability of human-caused ignitions because of the potential for ordnance-induced fires. Drier future conditions, coupled with a higher probability of human-caused ignition could present an increase in wildfire risk. Climate-driven changes in productivity could also influence carbon dynamics, especially in the context of management. To date, there is some empirical evidence to suggest that thinning reduces competition for water and that there can be differential effects to drought-resistance as a function of tree size. Kerhoulas et al. (2013) found that large ponderosa pines were less impacted by drought than small individuals of the same species following thinning. Given that thinning and burning treatments at all three installations result in more carbon being stored in larger individual trees, this result warrants further investigation.

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Appendix A

Appendix A contains parameter values used for LANDIS-II simulations and a list of references used to determine parameter values for each installation.

Table A1: Species-specific parameter values used for the species file at Fort Benning.

Species	Longevity (years)	Sexual Maturity (years)	Shade Tolerance (1-5)	Fire Tolerance (1-5)	Effective Seed Dispersal Distance (m)	Maximum Seed Dispersal Distance (m)
Pinus palustris	400	30	1	5	30	200
Pinus taeda	350	15	2	4	45	200
Pinus echinata	350	20	1	4	30	200
Quercus falcata	150	25	3	2	30	500
Quercus alba	300	20	3	3	30	500
Quercus marilandica	300	5	2	3	30	500
Carya tomentosa	300	25	3	1	30	500
Liquidambar styraciflua	350	20	2	1	60	180
Acer rubrum	250	10	4	1	100	1000
Quercus laevis	150	10	2	2	30	500

Species	Vegetative reproduction probability (0-1)	Minimum age of sprouting	Maximum age of sprouting	Post-fire Regeneration
Pinus palustris	0	0	3	None
Pinus taeda	0	0	3	None
Pinus echinata	0	0	25	Resprout
Quercus falcata	0.75	5	20	Resprout
Quercus alba	0.5	5	40	Resprout
Quercus marilandica	0.75	5	50	Resprout
Carya tomentosa	0.75	1	250	Resprout
Liquidambar styraciflua	0.75	5	50	Resprout
Acer rubrum	0.5	10	150	None
Quercus laevis	0.75	5	50	Resprout

Table A2: Species-specific parameter values for the Century Succession extension of LANDIS-II for Fort Benning.

Species	Functional Type	N Fixation	Growing Degree Days Min	Growing Degree Days Max	Minimum Jan Temp	Max Drought	Leaf Longevity	Epicormic Sprouting
Pinus palustris	1	N	4000	7000	3	0.423	2	N
Pinus taeda	1	N	4000	7000	1	0.360	3	N
Pinus echinata	1	N	4000	7000	1	0.423	3	N
Quercus falcata	2	N	4000	7000	-5	0.423	1	N
Quercus alba	2	N	3176	7000	1	0.330	1	N
Quercus marilandica	2	N	4000	7000	1	0.423	1	N
Carya tomentosa	2	N	3788	7000	1	0.300	1	N
Liquidambar styraciflua	2	N	3912	7000	1	0.300	1	N
Acer rubrum	2	N	1260	7000	-18	0.230	1	N
Quercus laevis	2	N	4000	7000	1	0.423	1	N

Species	Leaf Lignin	Fine Root Lignin	Wood Lignin	Coarse Root Lignin	Leaf C:N	Fine Root C:N	Wood C:N	Coarse Root C:N	Litter C:N
Pinus palustris	0.2	**0.35**	**0.35**	**0.35**	**50**	**50**	**380**	**170**	**100**
Pinus taeda	0.2	**0.35**	**0.35**	**0.35**	**50**	**50**	**380**	**170**	**100**
Pinus echinata	0.2	**0.35**	**0.35**	**0.35**	**50**	**50**	**380**	**170**	**100**
Quercus falcata	**0.293**	**0.23**	**0.23**	**0.35**	**24**	**48**	**500**	**333**	**55**
Quercus alba	**0.367**	**0.23**	**0.23**	**0.23**	**24**	**48**	**500**	**333**	**55**
Quercus marilandica	**0.293**	**0.23**	**0.23**	**0.35**	**24**	**48**	**500**	**333**	**55**
Carya tomentosa	**0.293**	**0.23**	**0.23**	**0.23**	**24**	**48**	**500**	**333**	**55**
Liquidambar styraciflua	**0.331**	**0.255**	**0.255**	**0.255**	**25**	**45**	**90**	**90**	**45**
Acer rubrum	**0.223**	**0.255**	**0.255**	**0.255**	**20**	**45**	**90**	**90**	**45**
Quercus laevis	**0.293**	**0.23**	**0.23**	**0.35**	**24**	**48**	**500**	**333**	**55**

Table A3: Functional group parameters for the Century Succession extension of LANDIS-II for Fort Benning.

Functional Group Name	Index	PPDF1 T-Mean	PPDF2 T-Max	PPDF3 T-Shape	PPDF4-T-Shape	FCFRAC Leaf	BTOLAI	KLAI	MAXLAI
Pine	1	28.0	45.0	4.5	5.0	0.37	0.00823	1000.0	10.0
Hardwood	2	27.0	45.0	3.0	3.5	0.5	0.00823	1000.0	20.0

Functional Group Name	Index	PPRPTS2	PPRPTS3	Wood Decay Rate	Monthly Wood Mortality	Mortality Age Shape	Leaf Drop Month
Pine	1	1.0	0.8	0.6	0.003	15	10
Hardwood	2	1.0	0.8	0.6	0.003	15	10

Table A4: Ecoregion parameters for the Century Succession extension of LANDIS-II for Fort Benning.

InitialEcoregionParameters										
Name	**SOM1**	**SOM1**	**SOM1**	**SOM1**	**SOM2**	**SOM2**	**SOM3**	**SOM3**	**Minrl**	
		C	**N**	**C**	**N**	**C**	**N**	**C**	**N**	**N**
		surf	**surf**	**soil**	**soil**					
Eco1		136.8	1.52	62.6	5.21	876.4	17.5	626.0	15.7	0.9
Eco2		98.4	1.09	124.2	10.4	1739.0	34.8	1242.1	31.1	0.9
Eco3		139.7	1.55	157.1	13.1	2199.4	44.0	1571.0	39.3	0.7
Eco4		139.7	1.55	31.5	2.62	441.4	8.8	315.3	7.88	0.7

	Soil Depth	% Clay	% Sand	Field Cap	Wilt Point	StormF Frac	BaseF Frac	Drain	Atm N dep	Atm N intercept	Latitude
Eco1	100	0.06	0.84	0.16	0.06	0.4	0.1	1.0	0.006	0.02	32
Eco2	100	0.12	0.66	0.23	0.10	0.4	0.1	1.0	0.006	0.02	32
Eco3	100	0.34	0.30	0.35	0.16	0.4	0.1	0.9	0.006	0.02	32
Eco4	100	0.36	0.53	0.32	0.17	0.4	0.1	0.9	0.006	0.02	32

Ecoregion Parameters cont	Decay Surf	Decay SOM1	Decay SOM2	Decay SOM3	Denitrifi
Eco1	0.127	0.305	0.0224	0.00008	0.03
Eco2	0.19	0.242	0.0134	0.00008	0.03
Eco3	0.134	0.332	0.013	0.00019	0.03
Eco4	0.15	1.08	0.0654	0.00058	0.03

Table A5: Species productivity parameters for the Century Succession extension of LANDIS-II for Fort Benning.

MonthlyMaxNPP (g m^{-2} $month^{-1}$)

	Eco1	Eco2	Eco3	Eco4
PIPA	110	125	150	150
PITA	110	125	150	150
PIEC	110	125	150	150
QUFA	100	100	120	120
QUAL	90	100	120	120
QUMA	90	100	110	110
CATO	90	100	120	120
LIST	80	90	140	140
ACRU	80	90	120	120
QULA	90	100	110	110

Maximum Biomass (g m^{-2})

	Eco1	Eco2	Eco3	Eco4
PIPA	21000	22000	24000	24000
PITA	21000	22000	24000	24000
PIEC	21000	22000	24000	24000
QUFA	21000	22000	22000	22000
QUAL	20000	22000	24000	24000
QUMA	20000	20000	21000	21000
CATO	20000	22000	24000	24000
LIST	15000	16000	24000	24000
ACRU	15000	16000	23000	23000
QULA	20000	20000	21000	21000

Table A6: Species-specific parameter values used for the species file for Camp Navajo.

Species	Longevity (years)	Sexual Maturity (years)	Shade Tolerance (1-5)	Fire Tolerance (1-5)	Effective Seed Dispersal Distance (m)	Maximum Seed Dispersal Distance (m)
Pinus ponderosa	400	7	2	4	35	120
Quercus gambelii	90	3	3	3	30	1000

Species	Vegetative reproduction probability (0-1)	Minimum age of sprouting	Maximum age of sprouting	Post-fire Regeneration
Pinus ponderosa	0	0	3	None
Quercus gambelii	0.75	1	80	resprout

Table A7: Species-specific parameter values for the Century Succession extension of LANDIS-II for Camp Navajo.

Species	Functional Type	N Fixation	Growing Degree Days Min	Growing Degree Days Max	Minimum Jan Temp	Max Drought	Leaf Longevity	Epicormic Sprouting
Pinus ponderosa	1	N	155	4000	-5	0.92	4.5	N
Quercus gambelii	2	N	800	4000	-5	0.90	1.0	Y

Species	Leaf Lignin	Fine Root Lignin	Wood Lignin	Coarse Root Lignin	Leaf C:N	Fine Root C:N	Wood C:N	Coarse Root C:N	Litter C:N
Pinus ponderosa	0.28	**0.2**	**0.25**	**0.25**	**48**	**48**	**250**	**170**	**100**
Quercus gambelii	0.175	**0.23**	**0.23**	**0.23**	**30**	**48**	**500**	**333**	**46**

Table A8: Functional group parameters for the Century Succession extension of LANDIS-II for Camp Navajo.

Functional Group Name	Index	PPDF1 T-Mean	PPDF2 T-Max	PPDF3 T-Shape	PPDF4- T-Shape	FCFRAC Leaf	BTOLAI	KLAI	MAXLAI
Pine	1	23.0	38.0	0.05	6.0	0.2	0.004	5000.0	10.0
Hardwood	2	23.0	35.0	0.05	7.0	0.3	0.004	5000.0	20.0

Functional Group Name	Index	PPRPTS2	PPRPTS3	Wood Decay Rate	Monthly Wood Mortality	Mortality Age Shape	Leaf Drop Month
Pine	1	1.0	0.5	0.4	0.002	10	10
Hardwood	2	1.0	0.5	0.4	0.002	10	10

Table A9: Ecoregion parameters for the Century Succession extension of LANDIS-II for Camp Navajo.

InitialEcoregionParameters									
Name	**SOM1 C surf**	**SOM1 N surf**	**SOM1 C soil**	**SOM1 N soil**	**SOM2 C**	**SOM2 N**	**SOM3 C**	**SOM3 N**	**Minrl N**
Eco1	412	4.5	90	7.5	2100	42	810	20	2.4
Eco2	412	4.5	90	7.5	2100	42	810	20	2.4
Eco3	412	4.5	90	7.5	2100	42	810	20	2.4
Eco4	412	4.5	90	7.5	2100	42	810	20	2.4
Eco5	412	4.5	90	7.5	2100	42	810	20	2.4
Eco6	412	4.5	90	7.5	2100	42	810	20	2.4

	Soil Depth	% Clay	% Sand	Field Cap	Wilt Point	StormF Frac	BaseF Frac	Drain	Atm N dep	Atm N intercept	Latitude
Eco1	100	0.13	0.55	0.24	0.09	0.3	0.6	0.7	0.035	0.004	35.2
Eco2	100	0.30	0.34	0.33	0.14	0.3	0.6	0.7	0.035	0.004	35.2
Eco3	100	0.30	0.34	0.33	0.14	0.3	0.6	0.7	0.035	0.004	35.2
Eco4	100	0.13	0.55	0.24	0.09	0.3	0.6	0.7	0.035	0.004	35.2
Eco5	100	0.30	0.34	0.33	0.14	0.3	0.6	0.7	0.035	0.004	35.2
Eco6	100	0.30	0.34	0.33	0.14	0.3	0.6	0.7	0.035	0.004	35.2

Ecoregion Parameters cont	Decay Surf	Decay SOM1	Decay SOM2	Decay SOM3	Denitrifi
Eco1	0.15	1.0	0.018	0.00035	0.1
Eco2	0.15	1.0	0.018	0.00035	0.1
Eco3	0.15	1.0	0.018	0.00035	0.1
Eco4	0.15	1.0	0.018	0.00035	0.1
Eco5	0.15	1.0	0.018	0.00035	0.1
Eco6	0.15	1.0	0.018	0.00035	0.1

Table A10: Species productivity parameters for the Century Succession extension of LANDIS-II for Camp Navajo.

MonthlyMaxNPP (g m^{-2} $month^{-1}$)

	Eco1	**Eco2**	**Eco3**	**Eco4**	**Eco5**	**Eco6**
PIPO	150	150	150	150	150	150
QUGA	75	75	75	75	75	75

Maximum Biomass (g m^{-2})

	Eco1	**Eco2**	**Eco3**	**Eco4**	**Eco5**	**Eco6**
PIPO	16000	16000	16000	16000	16000	16000
QUGA	10000	10000	10000	10000	10000	10000

Table A11: Species-specific parameter values used for the species file for JBLM.

Species	Longevity (years)	Sexual Maturity (years)	Shade Tolerance (1-5)	Fire Tolerance (1-5)	Effective Seed Dispersal Distance (m)	Maximum Seed Dispersal Distance (m)
Abies grandis	300	20	4	2	54	100
Acer macrophyllum	150	20	2	2	15	120
Alnus rubra	100	10	2	2	50	100
Fraxis latifolia	150	30	3	2	5	300
Pinus ponderosa	600	16	2	5	37	120
Pseudotsuga menziesii	750	25	3	3	100	1500
Quercus garryana	500	20	2	4	6	400
Thuja plicata	1000	25	5	2	100	122
Tsuga heterophylla	400	30	5	1	100	1600

Species	Vegetative reproduction probability (0-1)	Minimum age of sprouting	Maximum age of sprouting	Post-fire Regeneration
Abies grandis	0	0	0	None
Acer macrophyllum	0.7	0	100	resprout
Alnus rubra	0.7	0	10	resprout
Fraxis latifolia	0.7	0	100	resprout
Pinus ponderosa	0	0	3	None
Pseudotsuga menziesii	0	0	0	None
Quercus garryana	0.7	0	200	resprout
Thuja plicata	0.5	0	200	None
Tsuga heterophylla	0.3	0	2	None

Table A12: Species-specific parameter values for the Century Succession extension of LANDIS-II for JBLM.

Species	Functional Type	N Fixation	Growing Degree Days Min	Growing Degree Days Max	Minimum Jan Temp	Max Drought	Leaf Longevity	Epicormic Sprouting
Abies grandis	2	N	500	2450	-9	0.8	6	N
Acer macrophyllum	1	N	900	3100	-25	0.7	1	Y
Alnus rubra	1	Y	600	2200	-24	0.8	1	Y
Fraxis latifolia	1	N	150	2400	-22	0.7	1	N
Pinus ponderosa	2	N	800	3900	-41	0.9	4.5	N
Pseudotsuga menziesii	2	N	500	2500	-37	0.8	4.8	N
Quercus garryana	3	N	1400	2600	-34	0.9	1	Y
Thuja plicata	2	N	500	2000	-36	0.7	8.9	N
Tsuga heterophylla	2	N	500	1900	-31	0.7	1.6	N

Species	Leaf Lignin	Fine Root Lignin	Wood Lignin	Coarse Root Lignin	Leaf C:N	Fine Root C:N	Wood C:N	Coarse Root C:N	Litter C:N
Abies grandis	0.25	**0.22**	**0.35**	**0.35**	**42**	**27**	**500**	**170**	**77**
Acer macrophyllum	0.192	**0.224**	**0.25**	**0.26**	**20**	**30**	**440**	**90**	**62**
Alnus rubra	0.117	0.151	0.25	0.19	23	25	50	50	24
Fraxis latifolia	0.122	0.159	0.25	0.2	24	38	400	90	55
Pinus ponderosa	0.28	0.233	0.35	0.277	43	47	380	284	85
Pseudotsuga menziesii	0.155	0.296	0.269	0.323	42	52	455	214	68
Quercus garryana	0.176	0.22	0.14	0.26	32	63	63	62	33
Thuja plicata	0.18	0.205	0.293	0.245	53	29	80	38	100
Tsuga heterophylla	0.191	0.216	0.288	0.245	46	50	380	313	37

Table A13: Functional group parameters for the Century Succession extension of LANDIS-II for JBLM.

Functional Group Name	Index	PPDF1 T-Mean	PPDF2 T-Max	PPDF3 T-Shape	PPDF4- T-Shape	FCFRAC Leaf	BTOLAI	KLAI	MAXLAI
Hdwd_mesic	1	18.5	40.0	5.0	0.8	0.3	0.004	1000	4.0
Hdwd_dry	3	22.0	40.0	1.0	3.0	0.3	0.007	1000	4.0
Conifers	2	18.0	40.0	5.0	0.7	0.2	0.004	5000	12.0

Functional Group Name	Index	PPRPTS2	PPRPTS3	Wood Decay Rate	Monthly Wood Mortality	Mortality Age Shape	Leaf Drop Month
Hdwd_mesic	1	1.0	0.8	0.4	0.0024	10	10
Hdwd_dry	3	1.0	0.4	0.3	0.0024	15	10
Conifers	2	1.0	0.8	0.4	0.0015	15	10

Table A14: Ecoregion parameters for the Century Succession extension of LANDIS-II for JBLM.

Initial Ecoregion Parameters Name	SOM1 C Surf	SOM1 N Surf	SOM1 C Soil	SOM1 N Soil	SOM2 C	SOM2 N	SOM3 C	SOM3 N	Minrl N
Eco1	267	7	226.8	18.9	4158	207.9	3175.2	317.5	0.306
Eco2	2064	52	137.2	11.4	2514.6	125.7	1920.2	192	0.476

	Soil Depth	% Clay	% Sand	Field Cap	Wilt Point	StormF Frac	BaseF Frac	Drain	Atm N dep	Atm N intercept	Latitude
Eco1	100	0.035	0.823	0.069	0.034	0.01	0.14	0.9	0.0044	0.0343	47.0
Eco2	100	0.023	0.630	0.100	0.059	0.00	0.10	0.7	0.0044	0.0343	47.0

Ecoregion Parameters cont.	Decay Surf	Decay SOM1	Decay SOM2	Decay SOM3	Denitrifi
Eco1	0.3	0.2	0.025	0.00008	0.01
Eco2	0.3	0.7	0.060	0.00001	0.01

Table A15: Species productivity parameters for the Century Succession extension of LANDIS-II for JBLM.

MonthlyMaxNPP (g m^{-2} $month^{-1}$)

	Eco1	Eco2
Abies grandis	400	400
Acer macrophyllum	300	300
Alnus rubra	400	400
Fraxis latifolia	400	400
Pinus ponderosa	300	300
Pseudotsuga menziesii	350	350
Quercus garryana	200	200
Thuja plicata	300	300
Tsuga heterophylla	300	300

Maximum Biomass (g m^{-2})

	Eco1	Eco2
Abies grandis	50000	50000
Acer macrophyllum	50000	50000
Alnus rubra	50000	50000
Fraxis latifolia	50000	50000
Pinus ponderosa	60000	60000
Pseudotsuga menziesii	100000	100000
Quercus garryana	15000	15000
Thuja plicata	70000	70000
Tsuga heterophylla	100000	100000

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Appendix B

List of Scientific/Technical Publications that were produced during the course of this project.

Articles in peer-reviewed journals:

Martin, K.L., M.D. Hurteua, B.A. Hungate, G.W. Koch, M.P. North. 2015. Carbon tradeoffs of restoration and provision of endangered species habitat in a fire-maintained forest. Ecosystems 18, 76-88.

Hurteau, M.D. S. Liang, K.L. Martin, M.P. North, G.W. Koch, B.A. Hungate. In press. Restoring forest structure and process stabilizes forest carbon in wildfire-prone southwestern ponderosa pine. Ecological Applications, doi:10.1890/15-0337.1

Conference or symposium abstracts:

Hurteau, M.D. (Invited) Simulating climate, fire, and management influences on forest carbon dynamics in single- and multi-species forests of the Southwestern and Southeastern US. 2014 American Geophysical Union, Fall Meeting.

Laflower, D., M.D. Hurteau. Simulating carbon dynamics and species composition under projected changes in climate in the Puget Sound, Washington, USA. 2014 American Geophysical Union, Fall Meeting.

Hurteau, M.D., M.L. Brooks. (Invited) Short- and long-term effects of fire on carbon in US dry temperate forests. 2014 Southern Sierra Fire and Hydroclimate Workshop.

Martin, K.L., B.A. Hungate, G.W. Koch, M.P. North, M.D. Hurteau. Tradeoffs in forest carbon dynamics, fire management, and red-cockaded woodpecker habitat in longleaf pine ecosystems. 2013 meeting of the Ecological Society of America.

Colbert, C.T., K.L. Martin, M.D. Hurteau. Fire management impacts on carbon storage in Southwest ponderosa pine forests. 2013 meeting of the Ecological Society of America.

Hurteau, M.D. (Invited) Fire in a changing climate: quantifying and managing carbon trade-offs. Chicago Botanic Garden Research Symposium – Research-based answers to burning questions about the ecology of prescribed fires, October 26, 2012.

Martin, K.L., M.D. Hurteau, G.W. Koch, B.A. Hungate, M.P. North. Modeling the carbon implications of fire management in frequent-fire forests. RCN FORECAST New Investigators Conference, 2012.

Appendix C

User guides developed as part of this project:

Modeling the carbon implications of ecologically-based forest management, Camp Navajo Modeling Manual.

Modeling the carbon implications of ecologically-based forest management, Fort Benning Modeling Manual.

Modeling the carbon implications of ecologically-based forest management, Joint Base Lewis McChord Modeling Manual.