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Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems

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Abstract. Two forest management objectives being debated in the context of federally managed landscapes in the U.S. Pacific Northwest involve a perceived trade-off between fire restoration and carbon sequestration. The former strategy would reduce fuel (and therefore C) that has accumulated through a century of fire suppression and exclusion which has led to extreme fire risk in some areas. The latter strategy would manage forests for enhanced C sequestration as a method of reducing atmospheric CO_2 and associated threats from global climate change. We explored the trade-off between these two strategies by employing a forest ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on fire severity and the resulting long-term C dynamics among three Pacific Northwest ecosystems: the east Cascades ponderosa pine forests, the west Cascades western hemlock–Douglas-fir forests, and the Coast Range western hemlock–Sitka spruce forests. Our simulations indicate that fuel reduction treatments in these ecosystems consistently reduced fire severity. However, reducing the fraction by which C is lost in a wildfire requires the removal of a much greater amount of C, since most of the C stored in forest biomass (stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. For this reason, all of the fuel reduction treatments simulated for the west Cascades and Coast Range ecosystems as well as most of the treatments simulated for the east Cascades resulted in a reduced mean stand C storage. One suggested method of compensating for such losses in C storage is to utilize C harvested in fuel reduction treatments as biofuels. Our analysis indicates that this will not be an effective strategy in the west Cascades and Coast Range over the next 100 years. We suggest that forest management plans aimed solely at ameliorating increases in atmospheric CO_2 should forgo fuel reduction treatments in these ecosystems, with the possible exception of some east Cascades ponderosa pine stands with uncharacteristic levels of understory fuel accumulation. Balancing a demand for maximal landscape C storage with the demand for reduced wildfire severity will likely require treatments to be applied strategically throughout the landscape rather than indiscriminately treating all stands.

Key words: biofuels; carbon sequestration; fire ecology; fuel reduction treatment; Pacific Northwest, USA; Picea sitchensis; Pinus ponderosa; Pseudotsuga menziesii.

Introduction

Forests of the U.S. Pacific Northwest capture and store large amounts of atmospheric CO_2, and thus help mitigate the continuing climatic changes that result from extensive combustion of fossil fuels. However, wildfire is an integral component to these ecosystems and releases a substantial amount of CO_2 back to the atmosphere via biomass combustion. Some ecosystems have experienced an increase in the amount of CO_2 released due to a century-long policy of fire suppression that has led to increased levels of fuel buildup, resulting in wildfires of uncharacteristic severity. Fuel reduction treatments have been proposed to reduce wildfire severity, but like wildfire, these treatments also reduce the C stored in forests. Our work examines the effects of fuel reduction on wildfire severity and long-term C storage to gauge the strength of the potential trade-off between managing forests for increased C storage and reduced wildfire severity.

Forests have long been referenced as a potential sink for atmospheric CO_2 (Vitousek 1991, Turner et al. 1995, Harmon et al. 1996, Harmon 2001, Smithwick et al. 2002, Pacala and Socolow 2004), and are credited with contributing to much of the current C sink in the coterminous United States (Pacala et al. 2001, Hurtt et al. 2002). This U.S. carbon sink has been estimated to be between 0.30 and 0.58 Pg C/yr for the 1980s, of which between 0.17 Pg C/yr and 0.37 Pg C/yr has been attributed to accumulation by forest ecosystems (Pacala et al. 2001). While the presence of such a large sink has been valuable in mitigating global climate change, a substantial portion of it is due to the development of understory vegetation as a result of a national policy of fire suppression (Pacala et al. 2001, Donovan and Brown 2007). Fire suppression, while capable of incurring

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short-term climate change mitigation benefits by promoting the capture and storage of atmospheric CO_2 by understory vegetation and dead fuels (Houghton et al. 2000, Tilman et al. 2000), has, in part, led to increased and often extreme fire risk in some forests, notably *Pinus ponderosa* forests (Moeur et al. 2005, Donovan and Brown 2007).

Increased C storage usually results in an increased amount of C lost in a wildfire (Fahnestock and Agee 1983, Agee 1993). Many ecosystems show the effects of fire suppression (Schimel et al. 2001, Goodale et al. 2002, Taylor and Skinner 2003), and the potential effects of additional C storage on the severity of future wildfires is substantial. In the *Pinus ponderosa* forests of the east Cascades, for example, understory fuel development is thought to have propagated crown fires that have killed old-growth stands not normally subject to fires of high intensity (Moeur et al. 2005). Various fuel reduction treatments have been recommended for risk-prone forests, particularly a reduction in understory vegetation density, which can reduce the ladder fuels that promote such severe fires (Agee 2002, Brown et al. 2004, Agee and Skinner 2005). While a properly executed reduction in fuels could be successful in reducing forest fire severity and extent, such a treatment may be counterproductive to attempts at utilizing forests for the purpose of long-term C sequestration.

Pacific Northwest forests, particularly those that are on the west side of the Cascade mountain range, are adept at storing large amounts of C. Native long-lived conifers are able to maintain production during the rainy fall and winter months, thereby out-competing shorter-lived deciduous angiosperms with a lower biomass storage capacity (Waring and Franklin 1979). Total C storage potential, or upper bounds, of these ecosystems is estimated to be as high as 829.4 Mg C/ha and 1127.0 Mg C/ha for the western Cascades and Coast Range of Oregon, respectively (Smithwick et al. 2002). Of this high storage capacity for west Cascades and Coast Range forests, 432.8 Mg C/ha and 466.3 Mg C/ha, respectively, are stored in aboveground biomass (Smithwick et al. 2002), a substantial amount of fuel for wildfires.

High amounts of wildfire-caused C loss often reflect high amounts of forest fuel availability prior to the onset of fire. Given the magnitude of such losses, it is clear that the effect of wildfire severity on long-term C dynamics is central to our understanding of the global C cycle. What is not clear is the extent to which repeated fuel removals that are intended to reduce wildfire severity will likewise reduce long-term total ecosystem C storage (TECμ). Fuel reduction treatments require the removal of woody and detrital materials to reduce future wildfire severity. Such treatments can be effective in reducing future wildfire severity, but they likewise involve a reduction in stand-level C storage. If repeated fuel reduction treatments decrease the mean total ecosystem C storage by a quantity that is greater than the difference between the wildfire-caused C loss in an untreated stand and the wildfire-caused C loss in a treated stand, the ecosystem will not have been effectively managed for maximal long-term C storage.

Our goal was to test the extent to which a reduction in forest fuels will affect fire severity and long-term C storage by employing a test of such dynamics at multi-century time scales. Our questions were as follows: (1) To what degree will reductions in fuel load result in decreases in C stores at the stand level? (2) How much C must be removed to make a significant reduction in the amount of C lost in a wildfire? (3) Can forests be managed for both a reduction in fire severity and increased C sequestration, or are these goals mutually exclusive?

Methods

Model description

We conducted our study using an ecosystem simulation model, STANDCARB (Appendix A), that allows for the integration of many forest management practices as well as the ensuing gap dynamics that may result from such practices. STANDCARB is a forest ecosystem simulation model that acts as a hybrid between traditional single-life-form ecosystem models and multi-life-form gap models (Harmon and Marks 2002). The model integrates climate-driven growth and decomposition processes with species-specific rates of senescence and stochastic mortality while incorporating the dynamics of inter- and intraspecific competition that characterize forest gap dynamics. Inter- and intraspecific competition dynamics are accounted for by modeling species-specific responses to solar radiation as a function of each species' light compensation point as well as the amount of solar radiation delineated through the forest canopy to each individual. By incorporating these processes the model can simulate successional changes in population structure and community composition without neglecting the associated changes in ecosystem processes that result from species-specific rates of growth, senescence, mortality, and decomposition.

STANDCARB performs calculations on a monthly time step and can operate at a range of spatial scales by allowing a multi-cell grid to capture multiple spatial extents, as both the size of an individual cell and the number of cells in a given grid can be designated by the user. We used a 20×20 cell matrix for all simulations (400 cells total), with 15×15 m cells for forests of the west Cascades and Coast Range and 12×12 m cells for forests of the east Cascades. Each cell allows for interactions of four distinct vegetation layers, represented as upper canopy trees, lower canopy trees, a species-nonspecific shrub layer, and a species-nonspecific herb layer. Each respective vegetation layer can have up to seven live pools, eight detrital pools, and three stable C pools. For example, the upper and lower tree layers comprise seven live pools: foliage, fine roots, branches, sapwood, heartwood, coarse roots, and heart-rot, all of which are transferred to a detrital pool following mortality. Dead wood is separated into snags and logs to capture the effects of spatial position on microclimate. After detrital materials have undergone significant decomposition, they can contribute material to three increasingly decay-resistant, stable C pools: stable foliage, stable wood, and stable soil. Charcoal is created in both prescribed fires and wildfires and is thereafter placed in a separate pool with high decay resistance. Additional details on the STANDCARB model can be found in Appendix A.

Fire processes

We generated exponential random variables to assign the years of fire occurrence (sensu Van Wagner 1978) based on the literature estimates (see experimental design for citations) of mean fire return intervals (MFRI) for different regions in the U.S. Pacific Northwest. The cumulative distribution for our negative exponential function is given in Eq. 1 where X is a continuous random variable defined for all possible numbers x in the probability function P, and λ represents the inverse of the expected time $E[X]$ for a fire return interval given in Eq. 2:

$$P\{X \leq x\} = \int_0^x \lambda e^{-\lambda x} dx \tag{1}$$

where

$$E[X] = \frac{1}{\lambda}. \tag{2}$$

Fire severities in each year generated by this function are cell specific, as each cell is assigned a weighted fuel index calculated from fuel accumulation within that cell and the respective flammability of each fuel component, the latter of which is derived from estimates of wildfire-caused biomass consumption (see Fahnestock and Agee 1983, Covington and Sackett 1984, Agee 1993). Fires can increase (or decrease) in severity depending on how much the weighted fuel index of a given cell exceeds (or falls short of) the fuel level thresholds for each fire severity class (T_{light}, T_{medium}, T_{high}, and T_{max}), and the probability values for the increase or decrease in fire severity (P_{i} and P_{d}). For example, while the natural fire severity of many stands of the west Cascades can be described as high severity, other stands of the west Cascades have a natural fire severity that can be best described as being of medium severity (~60–80% overstory tree mortality) (Cissel et al. 1999). For these stands, medium-severity fires are scheduled to occur throughout the simulated stand and can increase to a high-severity fire depending on the extent to which the weighted fuel index in a cell exceeds the threshold for a high-severity fire, as greater differences between the fuel index and the fire severity threshold will increase the chance of a change in fire severity. Conversely, medium-severity fires may decrease to a low-severity fire if the fuel index is sufficiently below the threshold for a medium-severity fire. High-severity fires are likely to become medium-severity fires if the weighted fuel index within a given cell falls sufficiently short of the threshold for a high-severity fire, and low-severity fires are likely to become medium severity if the weighted fuel index in a given cell is sufficiently greater than the threshold for a medium-severity fire. Fuel level thresholds were set by monitoring fuel levels in a large series of simulation runs where fires were set at very short intervals to see how low fuel levels needed to be to create a significant decrease in expected fire severity. We note that, like fuel accumulation, the role of regional climate exerts significant influence on fire frequency and severity, and that our model does not attempt to directly model these effects. We suspect that an attempt to model the highly complex role of regional climate data on fine-scale fuel moisture, lightning-based fuel ignition, and wind-driven fire spread adds uncertainties into our model that might undermine the precision and applicability of our modeling exercise. For that reason we incorporated data from extensive fire history studies to approximate the dynamics of fire frequency and severity.

Final calculations for the expected stand fire severity $E[F_{\text{s}}]$ at each fire are performed as follows:

$$E[F_{\text{s}}] = \frac{100}{C}\sum_{i=1}^{n} c_{i(\text{L})}m_{i(\text{L})} + c_{i(\text{M})}m_{i(\text{M})} + c_{i(\text{H})}m_{i(\text{H})} \tag{3}$$

where C is the number of cells in the stand matrix and $c_{i(\text{L})}$, $c_{i(\text{M})}$, and $c_{i(\text{H})}$ are the number of cells with light, medium, and high-severity fires, and $m_{i(\text{L})}$, $m_{i(\text{M})}$, and $m_{i(\text{H})}$represent fixed mortality percentages for canopy tree species for light, medium, and high-severity fires, respectively. This calculation provides an approximation of the number of upper-canopy trees killed in the fire. The resulting expected fire severity calculation $E[F_{\text{s}}]$ is represented on a scale from 0 to 100, where a severity index of 100 indicates that all trees in the simulated stand were killed.

Our approach at modeling the effectiveness of fuel reduction treatments underscores an important trade-off between fuel reduction and long-term ecosystem C storage by incorporating the dynamics of snag creation and decomposition. Repeated fuel reduction treatments may result in a reduction in long-term C storage, but it is possible that if such treatments are effective in reducing tree mortality, they may also offset some of the C losses that would be incurred from the decomposition of snags that would be created in a wildfire of higher severity. STANDCARB accounts for these dynamics by directly linking expected fire severity with a fuel accumulation index that can be altered by fuel reduction treatments while also incorporating the decomposition of snags as well as the time required for each snag to fall following mortality.

Total ecosystem C storage (TEC) is calculated by summing all components of C (live, dead, and stable). For each replicate ($i = 1, 2, \ldots 5$) and for each period

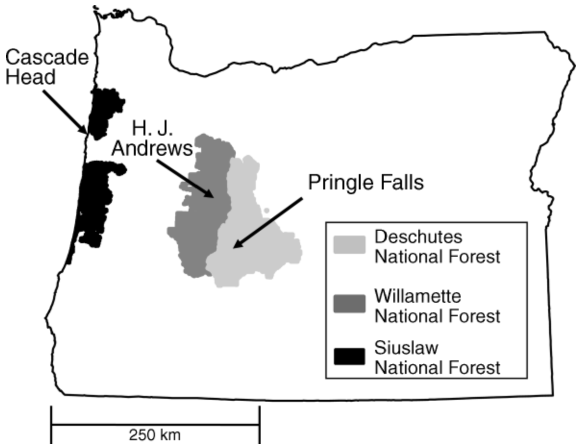

FIG. 1. Site locations in Oregon. Pringle Falls is our representative site for the east Cascades, H. J. Andrews is our representative site for the west Cascades, and Cascade Head is our representative site for the Coast Range.

between fires ($x = 1, 2, \dots P_i$), the mean total ecosystem C storage (TEC_μ) is calculated by averaging the yearly TEC values ($k = 1, 2, \dots R_x$).

$$TEC_{\mu(i,x)} = \frac{1}{R}\sum_{k=1}^{R} TEC_{(i,x,k)}.$$

Aggregating TEC_μ values in this manner permits the number of TEC_μ values to be the same as the number of $E[F_s]$ values, permitting a PerMANOVA analysis to be performed on $E[F_s]$ and TEC_μ.

Fuel reduction processes

STANDCARB's fire module allows for scheduled prescribed fires of a given severity (light, medium, high) to be simulated in addition to the nonscheduled wildfires generated from the aforementioned exponential random variable function. In addition to simulating the prescribed fire method of fuel reduction, STANDCARB has a harvest module that permits cell-by-cell harvest of trees in either the upper or lower canopy. This module allows the user to simulate understory removal or overstory thinning treatments on a cell-by-cell basis. Harvested materials can be left in the cell as detritus following cutting or can be removed from the forest, allowing the user to incorporate the residual biomass that results from harvesting practices. STANDCARB can also simulate the harvest of dead salvageable materials such as logs or snags that have not decomposed beyond the point of being salvageable.

Site descriptions

We chose the *Pinus ponderosa* stands of the Pringle Falls Experimental Forest as our representative for east Cascades forests (Youngblood et al. 2004). Topography in the east Cascades consists of gentle slopes, with soils derived from aerially deposited dacite pumice. The *Tsuga heterophylla–Pseudotsuga menziesii* stands of the H. J. Andrews Experimental Forest were chosen as our representative of west Cascades forests (Greenland 1994). Topography in the west Cascades consists of slope gradients that range from 20% to 60% with soils that are deep, well-drained dystrochrepts. The *Tsuga heterophylla–Picea sitchensis* stands of the Cascade Head Experimental Forest were chosen as our representative of Coast Range forests. We note that most of the Oregon Coast Range is actually composed of *Tsuga heterophylla–Pseudotsuga menziesii* community types, similar to much of the west Cascades. *Tsuga heterophylla–Picea sitchensis* communities occupy a narrow strip near the coast, due to their higher tolerance for salt spray, higher soil moisture optimum, and lower tolerance for drought compared to forests dominated by *Pseudotsuga menziesii* (Minore 1979), and we incorporate this region in order to gain insight into this highly productive ecosystem. Topography in the Cascade Head Experimental Forest consists of slope gradients of ~10% with soils that are silt loams to silt clay loams derived from marine siltstones. Site locations are shown in Fig. 1 and are located within three of the physiographic regions of Oregon and Washington as designated by Franklin and Dyrness (1988). Additional site data are shown in Table 1.

Experimental design

The effectiveness of forest fuel reduction treatments is often, if not always, inversely related to the time since their implementation. For this reason, our experiment incorporated a factorial blocking design where each ecosystem was subjected to four different frequencies of each fuel reduction treatment. We also recognize the fact that fire return intervals can exhibit substantial variation within a single watershed, particularly those with a high degree of topographic complexity (Agee 1993, Cissel et al. 1999), so we examined two likely fire regimes for each ecosystem. Historic fire return intervals may become unreliable predictors of future fire intervals (Westerling et al. 2006); thus ascertaining the differences in TEC_μ that result from two fire regimes might be a useful metric in gauging C dynamics resulting from fire regimes that may be further altered as a result of continued global climate change.

We based the expected fire return time in Eqs. 1 and 2 on historical fire data for our forests based on the following studies. Bork (1985) estimated a mean fire return interval of 16 years for the east Cascades *Pinus ponderosa* forests, and we also considered a mean fire return interval of 8 years for this system. Cissel et al. (1999) reported mean fire return intervals of 143 and 231 years for forests of medium- and high-severity (stand-replacing) fire regimes, respectively, among the *Tsuga heterophylla–Pseudotsuga menziesii* forests of the west Cascades. Less is known about the fire history of the Coast Range, which consists of *Tsuga heterophylla–Pseudotsuga menziesii* communities in the interior and *Tsuga heterophylla–Picea sitchensis* communities occu-

TABLE 1. Site characteristics (from Smithwick et al. 2002).

Site characteristic	Pringle Falls	H. J. Andrews	Cascade Head
Vegetation	PIPO	TSHE–PSME	TSHE–PISI
Elevation (m)	1359	785	287
Mean annual temperature (°C)	5.5	8.4	8.6
Mean annual precipitation (mm)	544	2001	2536
Soil porosity	sandy loam	loam	loam
Mean C storage potential (Mg C/ha)	183	829	1127

Note: Species codes: PIPO, *Pinus ponderosa*; TSHE, *Tsuga heterophylla*; PSME, *Pseudotsuga menziesii*; PISI, *Picea sitchensis*.

pying a narrow edge of land along the Oregon Coast. Work by Impara (1997) in the interior region of the Coast Range suggested a natural fire return interval (expected fire return time) of 271 years in the *Tsuga heterophylla–Pseudotsuga menziesii* zone, and Long et al. (1998) reported lake-derived charcoal sediment-based estimates of mean fire return interval for the Coast Range forests to be fairly similar, at 230 years. However, the *Tsuga heterophylla–Picea sitchensis* community type dominant in our study area of the Cascade Head Experimental Forest has little resistance to fire, and thus rarely provides a dendrochronological record. We estimated a mean fire return interval of 250 years as one fire return interval for a high-severity fire, derived from interior Coast Range natural fire return interval estimates, and also included another high-severity fire regime with a 500-year mean fire return interval in our analysis.

It is important to note that while the forests of the east Cascades exhibit a significant and visible legacy of effects from a policy of fire suppression, many of the mean fire return intervals for the forests of the west Cascades and Coast Range exceed the period of fire suppression ($\sim$100 years), and these forests in the west Cascades and Coast Range will not necessarily exhibit uncharacteristic levels of fuel accumulation (Brown et al. 2004). However, the potential lack of an uncharacteristic amount of fuel accumulation does not necessarily preclude these forests from future fuel reduction treatments or harvesting; thus we have included these possibilities in our analysis. The frequencies at which fuel reduction treatments are applied were designed to be reflective of literature-derived estimates of each ecosystem's mean fire return intervals, since forest management agencies are urged to perform fuel reduction treatments at a frequency reflective of the fire regimes and ecosystem-specific fuel levels (Franklin and Agee 2003, Dellasala et al. 2004). Treatment frequencies for the Coast Range and west Cascades were 100, 50, 25 years, plus an untreated control group, while treatment frequencies in the east Cascades were 25, 10, and 5 years, and an untreated control group.

We incorporated six different types of fuel reduction treatments largely based on those outlined in Agee (2002), Hessburg and Agee (2003), and Agee and Skinner (2005). Treatments 2–5 were taken directly from the authors' recommendations in these publications, treatment 1 was derived from the same principles used to formulate those recommendations, and treatment 6, clear-cutting, was not recommended in these publications but was incorporated into our analysis because it is a common practice in many Pacific Northwest forests. Treatments 1–4 were applied to all ecosystems, while treatments 5 and 6 were applied only to the west Cascades and Coast Range forests, as such treatments would be unrealistic at the treatment intervals necessary to reduce fire severity in the high-frequency fire regimes of the east Cascades *Pinus ponderosa* forests. Note that these treatments and combinations thereof are not necessarily utilized in each and every ecosystem. Managers of forests on the Oregon Coast, for example, would be unlikely to use prescribed fire as a fuel reduction technique. Our experimental design simply represents the range of all possible treatments that can be utilized for fuel reduction and is applied to all ecosystems purely for the sake of consistency.

1. *Salvage logging (SL)*.—The removal of large woody surface fuels limits the flame length of a wildfire that might enter the stand. Our method of ground fuel reduction entailed a removal of 75% of salvageable large woody materials in the stand. Our definition of salvage logging includes both standing and downed salvageable materials (sensu Lindenmayer and Noss 2006).

2. *Understory removal (UR)*.—Increasing the distance from surface fuels to flammable crown fuels will reduce the probability of canopy ignition. This objective can be accomplished through pruning, prescribed fire, or the removal of small trees. We simulated this treatment in STANDCARB by removing lower canopy trees in all cells.

3. *Prescribed fire (PF)*.—The reduction of surface fuels limits the flame length of a wildfire that might enter the stand. In the field, this is done by removing fuel through prescribed fire or pile burning, both of which reduce the potential magnitude of a wildfire by making it more difficult for a surface fire to ignite the canopy (Scott and Reinhardt 2001). We implemented this treatment in STANDCARB by simulating a prescribed fire at low severity for all cells.

4. *Understory removal and prescribed fire (UR + PF)*.—This treatment is a combination of treatments 2 and 3, where lower canopy trees were removed (treatment 2) before a prescribed fire (treatment 3) the following year for all cells.

5. *Understory removal, overstory thinning, and prescribed fire (UR + OT + PF).*—A reduction in crown density by thinning overstory trees can make crown fire spread less probable (Agee and Skinner 2005) and can reduce potential fuels by decreasing the amount of biomass available for accumulation on the forest floor. Some have suggested that such a treatment will be effective only if used in conjunction with UR and PF (Perry et al. 2004). We simulated this treatment in STANDCARB by removing all lower canopy trees (treatment 2), removing upper canopy trees in 50% of the cells, and then setting a prescribed fire (treatment 3) the following year. This treatment was excluded from the east Cascades forests because it would be unrealistic to apply it at intervals commensurate with the high-frequency fires endemic to that ecosystem.

6. *Understory removal, overstory removal, and prescribed fire (clear-cutting) (UR + OR + PF).*—Clear-cutting is a common silvicultural practice in the forests of the Pacific Northwest, notably on private lands in the Oregon Coast Range (Hobbs et al. 2002), and we included it in our analysis for two ecosystems (west Cascades and Coast Range) simply to gain insight into the effects of this practice on long-term C storage and wildfire severity. We simulated clear-cutting in STANDCARB by removing all upper and lower canopy trees, followed by a prescribed burn the following year. This treatment was excluded from the east Cascades forests because it would be unrealistic to apply it at intervals commensurate with the high-frequency fires endemic to that ecosystem.

7. *Control group.*—Control groups had no treatments performed on them. The only disturbances in these simulations were the same wildfires that occurred in every other simulation with the same MFRI.

In sum, our east Cascades analysis tested the effects of four fuel reduction treatment types, four treatment frequencies, including one control group, and two site mean fire return intervals (MFRI = 8 years, MFRI = 16 years). Our analysis of west Cascades and Coast Range forests tested the effects of six fuel reduction treatment types, four treatment frequencies, including one control group, and two site mean fire return intervals (MFRI = 143 years, MFRI = 230 years for the west Cascades, MFRI = 250 years, MFRI = 500 years for the Coast Range) on expected fire severity and long-term C dynamics. This design resulted in 32 combinations of treatment types for the east Cascades and 48 combinations of treatment types and frequencies for each fire regime in the west Cascades and Coast Range, with each treatment combination in each ecosystem replicated five times.

Biofuel considerations

Future increases in the efficiency of producing biofuels from woody materials may reduce potential trade-offs between managing forests for increased C storage and reduced wildfire severity. Much research is currently underway in the area of lignocellulase-based (as opposed to sugar- or corn-based) biofuels (Schubert 2006). If this area of research yields efficient methods of utilizing woody materials directly as an energy source or indirectly by converting them into biofuels such as ethanol, fuels removed from the forest could be utilized as an energy source and thus act as a substitute for fossil fuels by adding only atmosphere-derived CO_2 back to the atmosphere. However, the conversion of removed forest biomass into biofuels will only be a useful method of offsetting fossil fuel emissions if the amount of C stored in an unmanaged forest is less than the sum of managed stand TEC_{μ}, and the amount of fossil fuel emissions averted by converting removed forest biomass from a stand of identical size into biofuels over the time period considered. We performed an analysis on the extent to which fossil fuel CO_2 emissions can be avoided if we were to use harvested biomass directly for fuel or indirectly for ethanol production. We recognize that many variables need to be considered when calculating the conversion efficiencies of biomass to biofuels, such as the amount of energy required to harvest the materials, inefficiencies in the industrial conversion process, and the differences in efficiencies of various energy sources that exist even after differences in potential energy are accounted for. Rather than attempt to predict the energy expended to harvest the materials, the future of the efficiency of the industrial conversion process, and differences in energy efficiencies, we simply estimated the maximum possible conversion efficiency that can be achieved, given the energy content of these materials. The following procedure was used to estimate the extent to which fossil fuel CO_2 emissions can be avoided by substituting harvested biofuels as an energy source:

1) Estimate the mean annual biomass removal that results from intensive fuel reduction treatments.

2) Calculate the ratio of the amount of potential energy per unit C emissions for biofuels (both woody and ethanol) to the amount of energy per unit C emissions for fossil fuels.

3) Multiply the potential energy ratios by the mean annual quantity of biomass harvested to calculate the mean annual C offset by each biofuel type for each forest.

4) Calculate the number of years necessary for biofuels production to result in an offset of fossil fuel C emissions. This procedure was performed for two land-use histories: managed second-growth forests, and old-growth forests converted to managed second-growth forests.

Calculations for each ecosystem are shown in Appendix B.

Simulation spin-up

STANDCARB was calibrated to standardized silvicultural volume tables for Pacific Northwest stands. We then calibrated it to permanent study plot data from three experimental forests in the region (Fig. 1) to

TABLE 2. Treatment abbreviations.

Treatment abbreviation	Treatment
SL	salvage logging
UR	understory tree removal
PF	prescribed fire
UR + PF	understory tree removal + prescribed fire
UR + PF + OT	understory removal + prescribed fire + overstory thinning
UR + PF + OR	understory removal + prescribed fire + overstory removal

incorporate fuel legacies, which were taken from a 600-year spin-up simulation with fire occurrences generated from the exponential distribution in Eq. 1, where λ was based on each ecosystem's mean fire return interval. Spin-up simulations were run prior to the initiation of each series of fuel reduction treatments, and simulations were run for a total of 800 years for forests of the east Cascades, and a total of 1500 years for simulations of the west Cascades and Coast Range.

Data analysis

We employed a nonparametric multivariate analysis of variance, PerMANOVA (Anderson 2001), to test group-level differences in the effects of fuel reduction frequency and type on mean total ecosystem C storage and expected fire severity. PerMANOVA employs a test statistic for the F ratio that is similar to that of an ANOVA calculated using sum of squares, but unlike an ANOVA, PerMANOVA calculates sums of squares from distances among data points rather than from differences from the mean. PerMANOVA was used instead of a standard MANOVA because it was highly unlikely that our data would meet the assumptions of a parametric MANOVA. PerMANOVA analysis treated fuel reduction treatment type and treatment frequency as fixed factors within each respective fire regime for each ecosystem simulated. The null hypothesis of no treatment effect for different combinations of these factors on TEC_{μ} and $E[F_s]$ was tested by permuting the data into randomly assigned sample units for each combination of factors so that the number of replicates within each factor combination were fixed. Each of our 12 PerMANOVA tests incorporated 10 000 permutations using a Euclidian distance metric, and multiple pairwise comparison testing for differences among treatment types and treatment frequencies was performed when significant differences were detected (i.e., $P < 0.05$).

RESULTS

Results of the PerMANOVA tests indicate that mean expected fire severity ($E[F_s]$) and mean total ecosystem C storage (TEC_{μ}) were significantly affected by fuel reduction type ($P < 0.0001$), frequency ($P < 0.0001$), and interactions between type and frequency ($P < 0.0001$) in all three ecosystems. These results were significant for type, frequency, and interaction effects even when clear-cutting was excluded from the analysis for the west Cascades and Coast Range simulations, just as it was a priori for simulations of the east Cascades. When the PerMANOVA was performed on only one of our response variables ($E[F_s]$ or TEC_{μ}), groupwise comparisons of effects of treatment type showed that the most significant effects of treatment and frequency were related to TEC_{μ}. TEC_{μ} was strongly affected by treatment frequency for each fire regime in each ecosystem ($P < 0.0001$) and consistently showed an inverse relationship to the quantity of C removed in a given fuel reduction treatment, and was thus highly related to treatment type. $E[F_s]$, similar to TEC_{μ}, showed significant relationships with treatment frequency for all three ecosystems ($P < 0.0001$), with statistically significant differences among most treatment types. Boxplots of TEC_{μ} and $E[F_s]$ for each treatment type in each fire regime for each ecosystem are shown in Appendix C.

Fuel reduction treatments in east Cascades simulations reduced TEC_{μ} with the exception of one treatment type; UR treatments (see Table 2 for acronym descriptions) in these systems occasionally resulted in additional C storage compared to the control group. These differences were very small (0.6–1.2% increase in TEC_{μ}) but statistically significant (Student's paired t test, $P < 0.05$) for the treatment return interval of 10 years in the light fire severity regime No. 1 (MFRI = 8 years) and for all treatment return intervals in light fire severity regime No. 2 (MFRI = 16 years). The fuel reduction treatment that reduced TEC_{μ} the least was SL, which, depending on treatment frequency and fire regime, stored between 93% and 98% of the control group, indicating that there was little salvageable material. UR + PF, depending on treatment frequency and fire regime, resulted in the largest reduction of TEC_{μ} in east Cascades forests, storing between 69% and 93% of the control group.

Simulations of west Cascades and Coast Range forests showed a decrease in C storage for all treatment types and frequencies. Fuel reduction treatments with the smallest effect on TEC_{μ} were either SL or UR, which were nearly the same in effect. The treatment that most reduced TEC_{μ} was UR + OT + PF. Depending on treatment frequency and fire regime, this treatment resulted in C storage of between 50% and 82% of the control group for the west Cascades, and between 65% and 88% of the control group for the Coast Range. Simulations with clear-cutting (UR + OR + PF), depending on application frequency and fire regime, resulted in C storage that was between 22% and 58% of

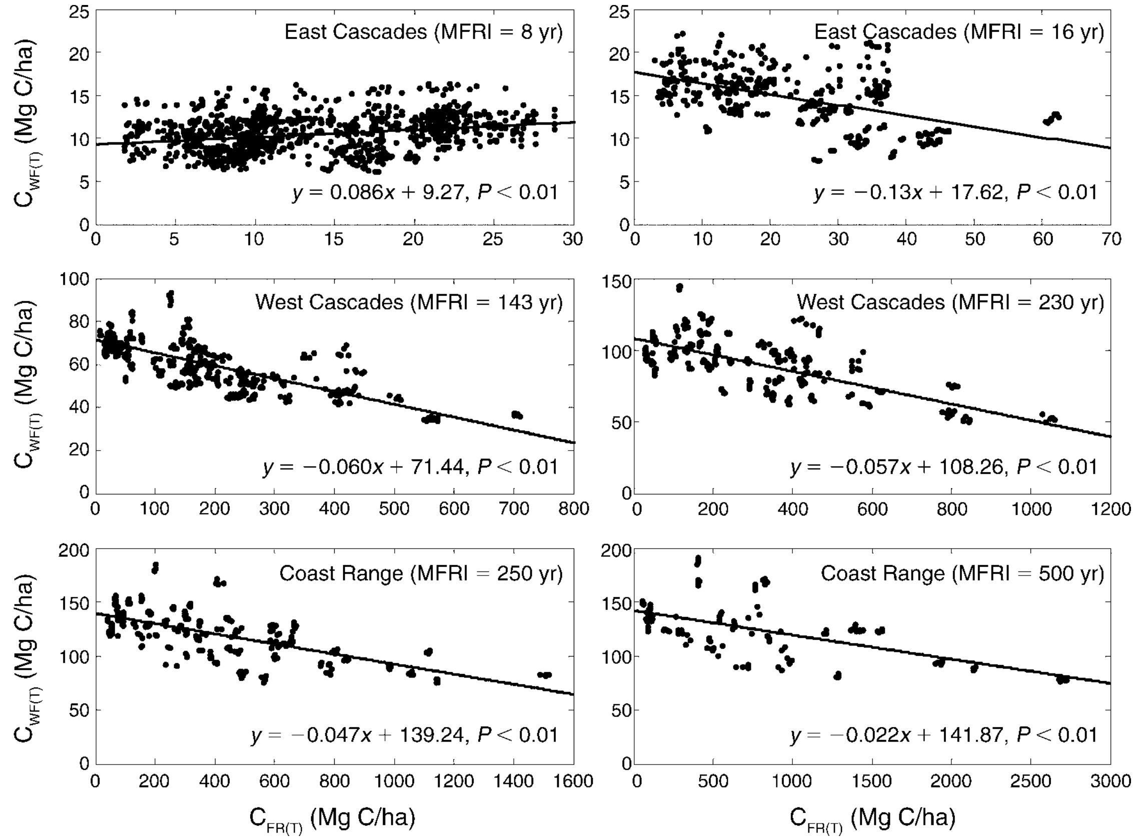

FIG. 2. Scatterplots of C removed in fuel reduction treatments between wildfires $C_{FR(T)}$ (representing fuel reduction [treatment]) and C lost in wildfires $C_{WF(T)}$ for the east Cascades, west Cascades, and Coast Range. Notice the differences in the axes scales. Also note the downward sloping trend for all ecosystems except for the east Cascades where MFRI = 8 years.

the control group for the west Cascades and between 44% and 87% of the control group for the Coast Range.

Similar to TEC_{μ}, $E[F_s]$ was significantly affected by fuel reduction treatments. Fuel reduction treatments were effective in reducing $E[F_s]$ for all simulations. UR treatments had the smallest effect on $E[F_s]$ in the east Cascades simulations and $E[F_s]$ in the east Cascades simulations was most affected by combined UR + PF treatments applied every five years, which reduced $E[F_s]$ by an average of 6.01 units (units range from 0 to 100, see Eq. 3) for stands with an MFRI = 8 years and by 11.08 units for stands with an MFRI = 16 years. In the west Cascades and Coast Range, $E[F_s]$ was least affected by UR treatments, similar to the east Cascades simulations. The most substantial reductions in $E[F_s]$ were exhibited by treatments that removed overstory as well as understory trees, as in treatments UR + OT + PF and UR + OR + PF. In the west Cascades simulations, depending on treatment frequency, $E[F_s]$ was reduced by an average of 11.72–15.68 units where the MFRI = 143 years and by an average of 3.92–26.42 units where the MFRI = 230 years when UR + OT + PF was applied. When UR + OT + PF was applied to the Coast Range, $E[F_s]$ was reduced by an average of 7.06–23.72 units where the MFRI = 250 years and by an average of 1.95–20.62 units where the MFRI = 500 years, depending on treatment frequency. Some UR + OR + PF treatments, when applied at a frequency of 25 years, resulted in $E[F_s]$ that was higher than that seen in UR + OT + PF in spite of lower TEC_{μ} in UR + OT + PF. A result such as this is most likely due to an increased presence of lower canopy tree fuels as a consequence of the increased lower stratum light availability that follows a clear-cut, as lower canopy tree fuels are among the highest weighted fuels in our simulated stands.

Modeled estimates of $E[F_s]$ were reflective of the mean amounts of C lost in a wildfire ($\bar{C}_{WF}$). $\bar{C}_{WF}$ was lower in the stands simulated with fuel reduction treatments compared to the control groups, with the exception of the east Cascades stands subjected to understory removal. Reductions in the amount of C lost in a wildfire, depending on treatment type and frequency, were as much as 50% in the east Cascades, 57% in the west Cascades, and 50% in the Coast Range. In the east Cascades simulations, amounts lost in wildfires were inversely related to the amounts of C removed in an average fire return interval for each ecosystem (Fig. 2), except for the Light Fire Regime No. 1 (MFRI = 8 years). Simulations in this fire regime revealed a slightly increasing amount of C lost in wildfires with increasing amounts removed, though amounts removed were nonetheless larger than the amounts lost in a typical wildfire.

Biofuels

Biofuels cannot offset the reductions in TEC_μ resulting from fuel reduction, at least not over the next 100 years. For example, our simulation results suggest that an undisturbed Coast Range *Tsuga heterophylla–Picea sitchensis* stand (where MFRI = 500 years) has a TEC_μ of 1089 Mg C/ha. By contrast, a Coast Range stand that is subjected to UR + OT + PF every 25 years has a TEC_μ of 757.30 Mg C/ha. Over a typical fire return interval of 450 years (estimated MFRI was 500 years, MFRI generated from the model was 450 years) this stand has 1107 Mg C/ha removed, a forest fuel/biomass production of 2.46 Mg $C{\cdot}ha^{-1}{\cdot}yr^{-1}$, which amounts to emissions of 1.92 Mg $C{\cdot}ha^{-1}{\cdot}yr^{-1}$ and 0.96 Mg $C{\cdot}ha^{-1}{\cdot}yr^{-1}$ that can be avoided by substituting biomass and ethanol, respectively, for fossil fuels (see calculations in Appendix B). This means that it would take 169 years for C offsets via solid woody biofuels and 339 years for C offsets via ethanol production before ecosystem processes result in net C storage offsets (see Fig. 3). Converting Coast Range old-growth forest to second-growth forest reduces the amount of time required for atmospheric C offsets to 34 years for biomass and 201 years for ethanol, and like all other biofuel calculations in our analysis, these are assuming a perfect conversion of potential energies. West Cascades *Tsuga heterophylla–Pseudotsuga menziesii* ecosystems (where MFRI = 230 years) that are subjected to UR + OT + PF every 25 years would require 228 years for C offsets using biomass as an offset of fossil-fuel-derived C and 459 years using ethanol. Converting west Cascades old-growth forest to second-growth forest reduces the amount of time required for atmospheric C offsets to 107 years for biomass fuels and 338 years for ethanol. Simulations of east Cascades *Pinus ponderosa* ecosystems had cases where stands treated with UR stored more C than control stands, implying that there is little or no trade-off in managing stands of the east Cascades for both fuel reduction and long-term C storage.

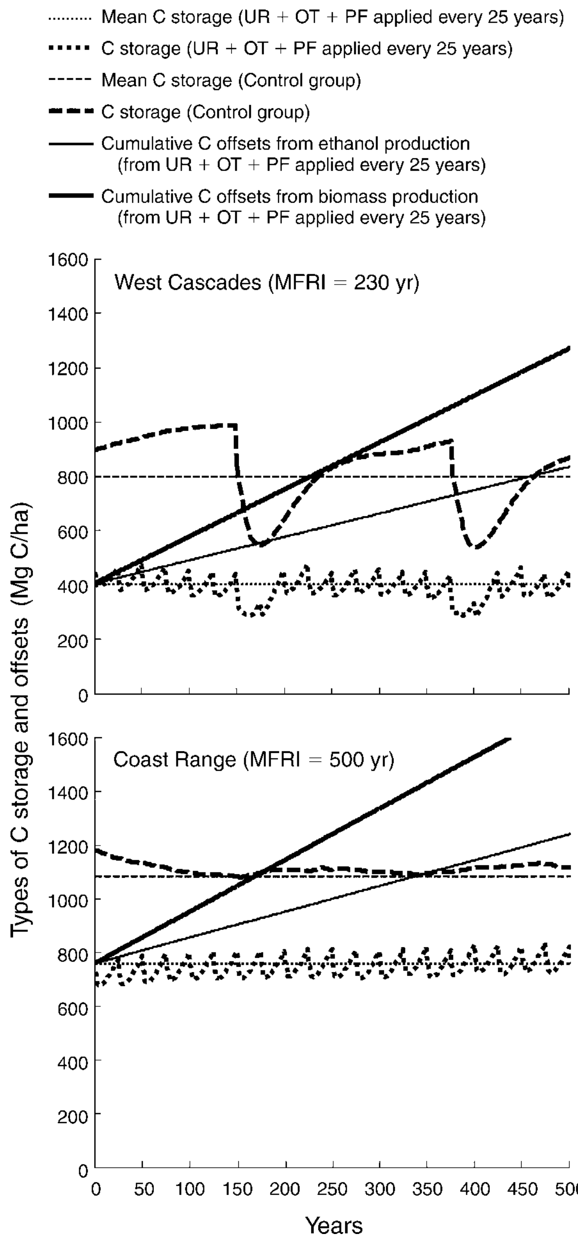

FIG. 3. Time series plots of C storage, mean C storage, and biofuels offsets for control groups and fuel reduction treatment UR + OT + PF (understory removal + overstory thinning + prescribed fire) applied to a second-growth forest every 25 years for the west Cascades and Coast Range. East Cascades simulations were excluded from this plot because there was little or no trade-off incurred in managing these forests for both fuel reduction and C sequestration.

Discussion

We employed an ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on expected fire severity and long-term C dynamics in three Pacific Northwest ecosystems: the *Pinus ponderosa* forests of the east Cascades, the *Tsuga heterophylla–Pseudotsuga menziesii* forests of the west Cascades, and the *Tsuga heterophylla–Picea sitchensis* forests of the Coast Range. Our fuel reduction treatments for east Cascades forests included salvage logging, understory removal, prescribed fire, and a combination of understory removal and prescribed fire. West Cascades and Coast Range simulations included these treatments as well as a combination of understory removal, overstory thinning, and prescribed fire. We also examined the effects of clear-cutting followed by prescribed fire on expected fire severity and long-term C storage in the west Cascades and Coast Range.

Our results suggest that fuel reduction treatments can be effective in reducing fire severity, a conclusion that is shared by some field studies (Stephens 1998, Pollet and

Omi 2002, Stephens and Moghaddas 2005) and modeling studies (Fulé et al. 2001). However, fuel removal almost always reduces C storage more than the additional C that a stand is able to store when made more resistant to wildfire. Leaves and leaf litter can and do have the majority of their biomass consumed in a high-severity wildfire, but most of the C stored in forest biomass (stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. For this reason, it is inefficient to remove large amounts of biomass to reduce the fraction by which other biomass components are consumed via combustion. Fuel reduction treatments that involve a removal of overstory biomass are, perhaps unsurprisingly, the most inefficient methods of reducing wildfire-related C losses because they remove large amounts of C for only a marginal reduction in expected fire severity. For example, total biomass removal from fuel reduction treatments over the course of a high-severity fire return interval (MFRI = 230 years) in the west Cascades could exceed 500 Mg C/ha while reducing wildfire-related forest biomass losses by only ~70 Mg C/ha in a given fire (Fig. 2). Coast Range forests could have as much as 2000 Mg C/ha removed over the course of an average fire return interval (MFRI = 500 years), only to reduce wildfire-related biomass combustion by ~80 Mg C/ha (Fig. 2).

East Cascades simulations also showed a trend of decreasing $E[F_s]$ with increasing biomass removal, though a higher TEC_μ was seen in some understory removal treatments compared to control groups. We believe that the removal of highly flammable understory vegetation led to a reduction in overall fire severity that consequently lowered overall biomass combustion, thereby allowing increased overall C storage. Such a result may be indicative of actual behavior under field conditions, but the very low magnitude of the differences between the treated groups and the control group (0.6%–1.2%) suggests caution in assuming that understory removal in this or any ecosystem can be effective in actually increasing long-term C storage. Furthermore, we recognize that the statistically significant differences between the treated and control groups are likely to overestimate the significance of the differences between groups that would occur in the field, as the differences we are detecting are modeled differences rather than differences in field-based estimates. Field-based estimates are more likely to exhibit higher inter- and intrasite variation than modeled estimates, even when modeled estimates incorporate stochastic processes, such as those in STANDCARB. Our general findings, however, are nonetheless consistent with many of the trends revealed by prior field-based research on the effects of fuel reduction on C storage (Tilman et al. 2000), though differences between modeled and field-based estimates are also undoubtedly apparent throughout other comparisons of treated and control stands in our study.

We note an additional difference that may exist between our modeled data and field conditions. Our study was meant to ascertain the long-term average C storage (TEC_μ) and expected fire severities ($E[F_s]$) for different fuel reduction treatment types and application frequencies, a goal not be confused with an assessment of exactly what treatments should be applied at the landscape level in the near future. Such a goal would require site-specific data on the patterns of fuel accumulation that have occurred in lieu of the policies and patterns of fire suppression that have been enacted in the forests of the Coast Range, west Cascades, and east Cascades for over a century. We did not incorporate the highly variable effects of a century-long policy of fire suppression on these ecosystems, as we know of no way to account for such effects in a way that can be usefully extrapolated for all stands in the landscape. *Pinus ponderosa* forests may exhibit the greatest amount of variability in this respect, as they are among the ecosystems that have been most significantly altered as a result of fire suppression (Veblen et al. 2000, Schoennagel et al. 2004, Moeur et al. 2005). Furthermore, additional differences may be present in our estimates of soil C storage for the east Cascades. Our estimates of soil C storage match up very closely with current estimates from the Pringle Falls Experimental Forest, but it is unclear how much our estimates would differ under different fuel reduction treatment types and frequencies. Many understory community types exist in east Cascades *Pinus ponderosa* forests (i.e., *Festuca idahoensis*, *Purshia tridentata*, *Agropyron spicatum*, *Stipa comata*, *Physocarpus malvaceus*, and *Symphoricarpos albus* communities) (Franklin and Dyrness 1988). An alteration of these communities may result from fuel reduction treatments such as understory removal or prescribed fire, leading to a change in the amount and composition of decomposing materials, which can influence long-term belowground C storage (Wardle 2002). Furthermore, there may be an increase in soil C storage resulting from the addition of charcoal to the soil C pool, whether from prescribed fire or wildfire (DeLuca and Aplet 2008).

By contrast, ecosystems with lengthy fire return intervals, such as those of the west Cascades and Coast Range, may not be strongly altered by such a policy, as many stands would not have accumulated uncharacteristic levels of fuel during a time of fire suppression that is substantially less than the mean fire return intervals for these systems. Forests such as these may actually have little or no need for fuel reduction due to their lengthy fire return intervals. Furthermore, fire severity in many forests may be more a function of severe weather events rather than fuel accumulation (Bessie and Johnson 1995, Brown et al. 2004, Schoennagel et al. 2004). Thus, the application of fuel reduction treatments such as understory removal is thought to be unnecessary in such forests and may provide only limited effectiveness (Agee and Huff 1986, Brown et al. 2004). Our results provide additional support for this notion, as they show a minimal effect of understory removal on expected fire severity in these forests, and if in fact climate has far stronger control over fire severity in these forests than fuel abundance, then the small reductions in expected fire severity that we have modeled for these fuel reduction treatments may be even smaller in reality.

We also note that the extent to which fuel reductions in these forests can result in a reduction in fire severity during the extreme climate conditions that lead to broad-scale catastrophic wildfires may be different from the effects shown by our modeling results, and are likely to be an area of significant uncertainty. Fuel reductions, especially overstory thinning treatments, can increase air temperatures near the ground and wind speeds throughout the forest canopy (van Wagtendonk 1996, Agee and Skinner 2005), potentially leading to an increase in fire severity that cannot be accounted for within our particular fire model. In addition to the microclimatic changes that may follow an overstory thinning, logging residues may be present on site following such a procedure, and may potentially nullify the effects of the fuel reduction treatment or may even lead to an increase in fire severity (Stephens 1998). Field-based increases in fire severity that occur in stands subjected to overstory thinning may in fact be an interaction between the fine fuels created by the thinning treatment and the accompanying changes in forest microclimate. These microclimate changes may lead to drier fuels and allow higher wind speeds throughout the stand (Raymond and Peterson 2005). While our model does incorporate the creation of logging residue that follows silvicultural thinning, increases in fire spread and intensity due to interactions between fine fuels and increased wind speed are neglected. However, we note that even if our model is failing to capture these dynamics, our general conclusion that fuel reduction results in a decrease in long-term C storage would then have even stronger support, since the fuel reduction would have caused C loss from the removal of biomass while also *increasing* the amount that is lost in a wildfire.

The amounts of C lost in fuel reduction treatments, whether nearly equal to or greater than our estimates, can be utilized in the production of biofuels. It is clear, however, that an attempt to substitute forest biomass for fossil fuels is not likely to be an effective forest management strategy for the next 100 years. Coast Range *Tsuga heterophylla–Picea sitchensis* ecosystems have some of the highest known amounts of biomass production and storage capacity, yet under the UR + OT + PF treatment a 169-year period is necessary to reach the point at which biomass production will offset C emitted from fossil fuels, and 338 years for ethanol production. Likewise, managed forests in the west Cascades require time scales that are too vast for biofuel alternatives to make a difference over the next 100 years. Even converting old-growth forests in these ecosystems would require at least 33 and 107 years for woody biomass utilization in the Coast Range and west Cascades, respectively, and these figures assume that all possible energy in these fuels can be utilized. Likewise, our ethanol calculations assumed that the maximum theoretical ethanol yield of biomass is realized, which has yet to be done (Schubert 2006); a 70% realization of our maximum yield is a more realistic approximation of contemporary capacities (Galbe and Zacchi 2002).

In addition to these lags, management constraints could preclude any attempt to fully utilize Pacific Northwest forests for their full biofuels production potential. Currently in the Pacific Northwest there are $\sim 3.6 \times 10^6$ ha of forests in need of fuel reduction treatments (Stephens and Ruth 2005), and in 2004 the annual treatment goal for this area was 52 000 ha (1.44%). Unless a significantly larger fuel reduction treatment workforce is employed, it would take 69 years to treat this area once, a period that approximates the effective duration of fire suppression (Stephens and Ruth 2005). The use of SPLATs (strategically placed area treatments) may be necessary to reduce the extent and effects of landscape-level fire (Finney 2001). SPLATs are a system of overlapping area fuel treatments designed to minimize the area burned by high-intensity head fires in diverse terrain. These treatments are costly, and estimates of such treatment costs may be underestimating the expense of fuel reduction in areas with high-density understory tree cohorts that are time consuming to extract and have little monetary value to aid in offsetting removal expenses (Stephens and Ruth 2005). Nevertheless, it is clear that not all of the Pacific Northwest forests that are in need of fuel reduction treatments can be reached, and the use of strategically placed fuel reduction treatments such as SPLATs may represent the best option for a cost-effective reduction in wildfire severity, particularly in areas near the wildland–urban interface. However, the application of strategically placed fuel reduction treatments is unlikely to be a sufficient means in itself toward ecosystem restoration in the forests of the east Cascades. Stand-level ecosystem restoration efforts such as understory removal and prescribed fire may need to be commenced once landscape-level reductions in fire spread risk have been implemented.

Conclusions

Managing forests for the future is a complex issue that necessitates the consideration of multiple spatial and temporal scales and multiple management goals. We explored the trade-offs for managing forests for fuel reduction vs. C storage using an ecosystem simulation model capable of simulating many types of forest management practices. With the possible exception of some xeric ecosystems in the east Cascades, our work suggests that fuel reduction treatments should be forgone if forest ecosystems are to provide maximal amelioration of atmospheric CO_2 over the next 100 years. Much remains to be learned about the effects of forest fuel reduction treatments on fire severity, but our results demonstrate that if fuel reduction treatments are effective in reducing fire severities in the western hemlock–Douglas-fir forests of the west Cascades and the western hemlock–Sitka spruce forests of the Coast Range, it will come at the cost of long-term C storage, even if harvested materials are utilized as biofuels. We agree with the policy recommendations of Stephens and Ruth (2005) that the application of fuel reduction treatments may be essential for ecosystem restoration in forests with uncharacteristic levels of fuel buildup, as is often the case in the xeric forest ecosystems of the east Cascades. However, this is often impractical and may even be counterproductive in ecosystems that do not exhibit uncharacteristic or undesirable levels of fuel accumulation. Ecosystems such as the western hemlock–Douglas-fir forests in the west Cascades and the western hemlock–Sitka spruce forests of the Coast Range may in fact have little sensitivity to forest fuel reduction treatments and may be best utilized for their high C sequestration capacities.

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APPENDIX A

STANDCARB model description (*Ecological Archives* A019-028-A1).

APPENDIX B

Biofuels analysis calculations (*Ecological Archives* A019-028-A2).

APPENDIX C

Carbon storage and fire severity results for each treatment type and frequency (*Ecological Archives* A019-028-A3).