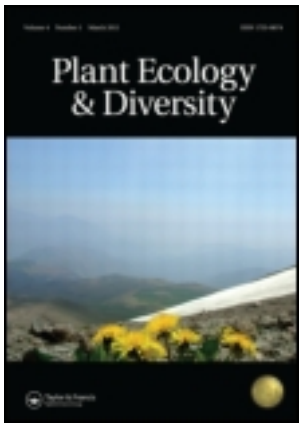

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Thinning effects on forest productivity: consequences of preserving old forests and mitigating impacts of fire and drought

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Thinning effects on forest productivity: consequences of preserving old forests and mitigating impacts of fire and drought

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Background: Management strategies have been proposed to minimise the effects of climate change on forest resilience.
Aims: We investigated the Pacific Northwest US region forest carbon balance under current practices, and changes that may result from management practices proposed for the region's 34 million ha of forests to mitigate climate change effects.
Methods: We examined the relationship between net primary production (NPP) and biomass, using plot data, and estimated the effects of proposed clear-cut harvest of young mesic forests for wood products and bioenergy while preserving mesic mature/old forests for biodiversity (Sparing), thinning all forests (Sharing) and a combination of sparing mesic mature and old, clearing mesic young and thinning dry forests (Sparing/Sharing).
Results: The forests of the region were found highly productive (NPP 163 Tg C year^{-1}) and a strong carbon sink with net ecosystem production of 45 Tg C year^{-1}. Observations indicated the relationship between NPP and biomass was not significantly different for thinned versus unthinned stands, after accounting for site quality and precipitation effects. After simulating proposed management to mitigate climate change, regional NPP was reduced by 35% (Sparing), 9% (Sharing) and 29% (Sparing/Sharing) compared with current practices.
Conclusions: Applying management practices appropriate for current forest conditions to mitigate future climate change impacts can be accomplished, but at a cost of reducing NPP. Sparing all forests >50 years old resulted in the largest NPP reduction, but the impact could be reduced by clearing only a subset of young forests.

Keywords: disturbance; drought; fire; forest carbon processes; harvest

Introduction

Climate change is expected to include warming, changes in precipitation regimes and lengthen forest growing seasons which can exacerbate drought stress and contribute to disturbance from insects (Kurz et al. 2008a, 2008b), pathogens and wildfire (Westerling et al. 2006). These disturbance agents often occur in sequence, further complicating understanding of potential trajectories of change in carbon cycling. In addition, natural disturbances can be amplified by anthropogenic activities, increasing the vulnerability of forests (Raffa et al. 2008). There is much uncertainty about how and where to mitigate these effects at the local to regional scales. A question is, if forests are thinned to minimise the effect of potential drought and fire, how does this affect forest carbon stocks, productivity and vulnerability to mortality?

Natural disturbances affect forest carbon dynamics for years to decades. Recent large-scale events such as hot/dry years across Europe and the USA, and large-scale forest mortality from insects in Canada have given us a window to the future on terrestrial ecosystem responses. After a drought, the effects on moisture reserves, soil nutrients and plant carbohydrates lead to longer-term effects in plant carbon cycling, and potentially mortality. Photosynthesis, respiration and net ecosystem production (NEP, net primary production (NPP) minus heterotrophic respiration) decline in most cases in drought years (Ciais et al. 2005; Reichstein et al. 2007), and carry-over effects of multiple years of drought can lead to depressed carbon uptake in subsequent years (Thomas et al. 2009). Declines in NEP following major disturbances can result in the forest becoming a net carbon source for an average of 15–20 years until productivity increases again and decomposition of dead material decreases (Luyssaert et al. 2008; Amiro et al. 2010). Direct and carry-over effects, mortality and consequently species competition in response to drought are strongly related to the survival strategies of species (van der Molen et al. 2011).

Previous studies in the Pacific Northwest US region suggest that in semi-arid regions, old forests can respond positively to thinning treatments to alleviate drought stress (Kolb et al. 2007). In one ponderosa pine study, basal area increment of individual trees increased two to threefold 5 years after thinning, water stress was reduced compared with unthinned trees, and this was sustained for up to 15 years after the basal area was reduced by 60–80% (McDowell et al. 2003). Thus, it is reasonable to suggest that thinning of semi-arid forests that are at high risk of crown fires and drought effects could improve sustainability of semi-arid forests.

Forest thinning and other management practices are thought to reduce mortality and increase growth, and have

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therefore been proposed and implemented to minimise the effects of drought, warmer temperatures and longer growing seasons expected with climate change. However, inventory analysis, field studies and simulations indicate that thinning forests to increase climate adaptability or alter fire behaviour would increase forest carbon emissions and losses (Campbell et al. 2007; Mitchell et al. 2009; Hudiburg et al. 2011). In addition, it is proposed that mature (>50 years) and old forests should be protected for biodiversity and natural adaptation to climate change (e.g. landscape connectivity for species migration), while continuing to harvest younger stands for wood products and bioenergy as an alternative energy source. Forest resilience and sustainability are paramount, and the carbon consequences of such mitigation activities need to be determined.

Given that several functions (i.e. biodiversity conservation, adaptability to climate change, mitigating climate change through carbon sequestration and mitigating the effects of climate change on forests through decreasing stand susceptibility to fire and drought stress) need to be simultaneously realised, implementation of forest management needs to account for large-scale considerations. One such large-scale consideration is whether these functions are best realised through land sharing or land sparing. Land sharing is a management strategy that would aim to integrate biodiversity conservation with forest production on the same land by using wildlife-friendly methods. Land sparing, on the other hand, would separate land for conservation (e.g. preserves) from land used for production (e.g. intensive plantations; Phalan et al. 2011).

Our goal was to examine changes in productivity that could result from typical land sharing and land sparing harvest practices proposed for Pacific Northwest US forests to mitigate climate change impacts on forests. Our objectives were to examine (1) the regional NPP and NEP that are influenced by historic and current management practices; (2) the effect of precipitation, forest age and thinning on the relationship between NPP and biomass within ecoregions using experimental and inventory data; and (3) the regional effect on NPP of: (a) clear-cut harvesting young forests (on a 50-year cycle, converting a portion of the landscape to short rotation) and setting aside mature and old forests (i.e. land sparing forests >50 years) in all ecoregions; (b) thinning all ecoregions at a moderate level (i.e. land sharing); and (c) sparing mesic mature/old, clear-cutting mesic young and thinning all dry forests (<650 mm annual precipitation) to reduce fire and drought stress in the latter (Sparing/Sharing).

Materials and methods

Study area

We conducted this study in forests of the Pacific Northwest US region, which consists of the states of Washington (WA), Oregon (OR) and northern California (CA). California and Oregon have a similar amount of forested area, with 12.8 and 12.2 × 10^6 hectares, respectively, and Washington has 9.0 × 10^6 forested hectares, for a regional total of 34 million hectares. Wilderness areas and reserves were excluded from the analysis, which partly explains the differences in forest area, as Washington has more protected areas.

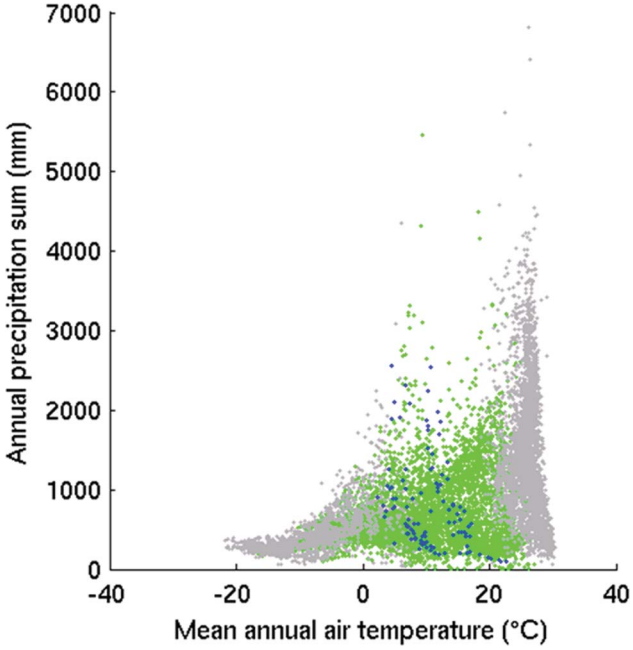

Figure 1. Regional climate range and variability of mean annual temperature (°C) and annual precipitation sum (mm). Light grey dots represent global temperature and precipitation regimes, darker grey dots represent temperate climates and the darkest dots represent the forest type averages across the study region.

There is a strong climatic and vegetation gradient from the mild mesic coastal forests (mean annual precipitation 2500 mm year^{-1}) to the semi-arid pine and cold desert juniper woodlands (300 mm year^{-1}; Figure 1). Primary species in the mesic area are Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco), western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), Sitka spruce (*Picea sitchensis* (Bong.) Carr.) and coastal redwood (*Sequoia sempervirens* (D. Don) Endl.) (Franklin and Halpern 2000). These forests have the potential to live for over 1000 years, and the mesic ecoregions have some of the highest biomass accumulation and productivity levels in the world (Hudiburg et al. 2009; Keith et al. 2009).

Data sources

Our primary data sources were the Forest Inventory and Analysis (FIA) dataset, plot data from our own productivity and thinning studies, and remote sensing data products. FIA data are collected annually on all types of forest land across the US. The FIA inventory has a probability-based design consisting of 0.404 ha plots systematically gridded across the landscape, encompassing a representative range of stand ages, disturbance histories, ownerships and land cover types. The FIA data were combined with Landsat-based mapping of vegetation type, fire and fuel characteristics, and 200 supplementary plots (Sun et al. 2004; Hudiburg et al. 2009) to produce mapped estimates

of current total forest NPP and NEP over the Pacific Northwest US region (details of computation methods in Hudiburg et al. 2011). Forest NPP included all trees regardless of stature, and understory shrubs. Wood NPP (bole, bark, branches and coarse root) was estimated as the difference between the biomass of each component at current and previous time steps using tree increment core data. Foliar NPP was estimated as the foliage biomass divided by the average leaf retention time using species-specific look-up tables constructed for the supplemental plot data. Fine root NPP was estimated as fine root biomass multiplied by average fine root turnover. We defined NEP as the difference between annual NPP and heterotrophic respiration. While direct measurements of soil respiration were not available on FIA plots, we were able to calculate NEP using a mass-balance approach and supplementary plot data where soil and root carbon pools were measured (Hudiburg et al. 2011; Campbell et al. 2009):

$$NEP = \text{Above-ground NPP} - \text{dead wood decomposition} - \text{litterfall} + \Delta\text{root} + \Delta\text{soil C}. \quad (1)$$

Plot means of current NPP and NEP were scaled to regional and state totals, using spatially explicit forest cover, ecoregion and succession class data products available in 30 m × 30 m resolution from LandFire Landsat-derived products (USGS 2009). Ecoregions denote areas within which ecosystem characteristics are generally similar (geology, physiography, vegetation, climate, soils, land use, wildlife and hydrology; Omernik 1987, 2004). There are 18 ecoregions in the Pacific Northwest US region (Table 1). The remote sensing product LandFire has five succession classes associated with vegetation development (A, early development, post-replacement; B, mid-development closed; C, mid-development open; D, late development closed; E, late development open). We chose to use pixels labelled as 'A' or 'B' for selection of areas which were considered less than ca. 50 years. Plot values were aggregated by climatic region (ecoregion), forest type and age class (succession class), and this look-up table was used to assign a value to each associated 30 m pixel.

We examined the effects of thinning on the relationship between NPP and biomass using plot-level data from thinning experiments and inventories. The effects of thinning on NPP have often been studied as a function of stand age. Where such an approach may be appropriate for plantation-type management or stand-level studies, it can be challenging for large-scale studies or to compare stands under different management strategies because thinning can occur at different stand ages. For this reason, we studied the effect of thinning on the relationship between NPP and stand biomass. Because the relationship is also affected by forest type, precipitation and site quality, we included these variables in our analysis. This information was used to prescribe growth after thinning in the regional analysis.

One thinning experiment was in the Northern Sierras of California, where ponderosa pine (*Pinus ponderosa* Douglas ex C. Lawson) was replanted in 1961 after stand-replacing fire. The area experiences average annual air temperature of 12 °C, and precipitation of 950 mm, most of which occurs outside the growing season. After the fire, the area was harvested, root-raked and windrowed into debris piles, and then planted. The stands were thinned once from below, removing 50% of the basal area, resulting in reduction of basal area from 40 to 20 $m^2\,ha^{-1}$ (Campbell et al. 2009). The thinning regime applied was not atypical of fuel-reduction practices in mature ponderosa pine in the region. We examined the relation between total NPP and biomass of the untreated stands ($n = 4$) with those 3 years after a single thinning treatment ($n = 5$) and those 16 years after a single thinning treatment ($n = 5$). All stands were measured at the same time and were ca. 30 years old.

We also analysed data from observational plots, thinned and unthinned mature (40–80 years) ponderosa pine stands in the Metolius, Oregon watershed, a drier region with 30-year mean annual precipitation of 360 mm (Law et al. 2001). These studies allowed us to determine if the relationship between NPP and biomass changed for an extended period (ca. 10 years) after thinning in pine forests subject to summer drought but with different annual precipitation, as is expected for mitigating drought effects. In addition, we examined the relationship between NPP and biomass for thinned and unthinned plots within ecoregions (Omernik 1987) using inventory data after analysis of the influence of precipitation and age.

Projecting post-treatment C-balances

We aimed to project post-treatment NPP and compare it with that of current practices across the Pacific Northwest US region to examine the change in productivity associated with proposed management actions to mitigate climate change impacts on forests. Thinning treatments specific to mesic and dry forest were applied on the current forests to estimate their future biomass and NPP. Over 8000 inventory plots in all forest types across the whole region were virtually thinned according to stand density reduction requirements, and the new biomass values were calculated. The aforementioned relationships between NPP and biomass, accounting for forest type and ownership (site quality) and precipitation, were then applied to estimate future NPP.

The basal area removals, maximum bole size and areas treated were determined for three treatment levels: (1) clear-cut harvest young forests (on a 50-year cycle, converting a portion of the landscape to short-rotation) and set aside mature and old forests (i.e. Sparing forests > 50 years old) in all ecoregions; (2) thinning all ecoregions at moderate level (i.e. Sharing); and (3) spare mesic mature and old forests, clear-cut mesic young forests and thin all dry forests (< 650 mm annual precipitation) (i.e. Sparing/Sharing). The Sparing treatment essentially converts most previously harvested lands into short-rotation forests, and protects mature and old mesic forests for

Table 1. Ecoregion characteristics including dominant forest types, total area, total number of plots and number of thinned plots, mean annual precipitation (MAP), and regressions for predictions of NPP from biomass (B), forest type (FT), precipitation (P) and ownership (O); organised by MAP (high to low). Regressions were conducted for each ecoregion separately.

Ecoregion[1] (mean forest age)	Forest (ha)	Total plots, thinned plots	Dominant forest types	MAP	Parameters ($r^{2)}$)
CR (62)	4812627	1008, 127	Douglas-fir, Sitka Spruce, Redwood, Western Red Cedar, Fir-hemlock	1742	B + O(0.62)
WC (105)	4329871	987, 63	Douglas-fir, Hemlock, Mixed Conifer, Red Fir, Western Red Cedar	1688	B + P + O(0.56)
KM (105)	3748465	1103, 68	Mixed Conifer, Mixed Evergreen, Red Fir, Douglas-fir, Riparian, Oak	1549	B + FT + O(0.57)
NC (104)	2311424	452, 32	Fir-Hemlock, Mixed Conifer, Spruce-Fir, Western Red Cedar, Riparian	1548	B + FT + P + O(0.58)
PL (50)	1102015	164, 0	Douglas-fir, Riparian, Western Red Cedar, Sitka Spruce	1304	B + FT (0.73)
WV (45)	538681	105, 0	Douglas-fir, Hemlock, Riparian	1280	B + FT (0.68)
SM (113)	730051	167, 0	Mixed Evergreen, Mixed Conifer, Mixed Oak, Oak Woodland, Riparian	1064	B + FT + P(0.52)
SN (116)	1022645	1215, 124	Mixed Conifer, Red Fir, Ponderosa Pine, Mixed Oak-Conifer, Pine	915	B + FT + P + O(0.53)
CO (97)	2688165	447, 19	Pine, Mixed Conifer, Redwood, Oak Woodland and Savanna	652	B + FT + P(0.69)
EC (92)	3545116	1032, 144	Ponderosa Pine, Mixed Conifer, Juniper, Pine, Red Fir	630	B + FT + P + O(0.62)
NR (73)	1514359	329, 50	Mixed conifer, Riparian, Spruce-Fir, Ponderosa Pine	613	B + FT + P(0.70)
BM (93)	3312268	827, 62	Mixed Conifer, Ponderosa Pine, Juniper, Spruce-Fir	552	B + FT + O(0.72)
CB (135)	352650	105, 0	Pinyon-Juniper, Ponderosa Pine, Pine	445	B + P (0.56)
CV (na)	170243	na	Oak, Pine, Riparian, Salt Desert Scrub, Mixed Oak Savanna	412	Na
CP (67)	253667	63, 13	Mixed Conifer, Ponderosa Pine, Riparian	330	B + O(0.60)
NB (115)	478106	95, 3	Juniper, Aspen, Pinyon-Juniper, Ponderosa Pine, Mountain Mahogany	304	B (0.42)
MB (122)	93889	35, 0	Pinyon-Juniper, Mixed Oak Woodland	185	B + FT (0.47)
SB (na)	2175	na	Pinyon-Juniper	110	Na

[1]BM, Blue Mountains; CB, Central Basin; CO, California Chaparral and Oak Woodlands; CP, Columbia Plateau; CR, Coast Range; CV, Central California Valley; EC, East Cascades; KM, Klamath Mountains; MB, Mohave Basin; NB, North Basin and Range; NC, North Cascades; NR, Northern Rockies; PL, Puget Lowlands; SB, Sonoran Basin; SM, Southern California Mountains; SN, Sierra Nevada; SR, Snake River; WC, West Cascades; WV, Willamette Valley.

biodiversity and other ecosystem values. The thinning treatment in dry ecoregions (dry portion of Sharing and Sharing/Sparing treatments) removes smaller trees and some larger trees to provide at least 9 Mg of dry biomass ha^{-1} (4.5 Mg C ha^{-1}) of merchantable biomass (Skog et al. 2008), with the expectation that the merchantable biomass would help pay for removal of small trees that are potential fuel ladders to the crowns of larger trees. All treatments exclude public forest reserves. A treatment period of 20 years was assumed to be the amount of time required to treat the entire landscape in the Sharing treatment. Pixels were harvested over a 20-year period, so that only 5% of the treatable area was treated each year in all three scenarios. It is also a common timeframe of policy actions and verification of expected results. FIA plots with stand densities greater than 300 trees ha^{-1} and located on forestland capable of producing 10 Mg of merchantable wood ha^{-1} year^{-1} were thinned according to each treatment. New plot mean biomass values were scaled to state and ecoregion boundaries to determine the removal totals.

Biomass removal levels were defined by current or proposed practices (USDA 2010), and treatments were designed to reduce crown fire potential by thinning from below (Stephens et al. 2009). Synthesis of fuel treatment studies showed that stand basal area was reduced by an average of 48% (Evans and Finkral 2009); however, Johnson et al. (2007) indicated that 30–55% basal area removal plus surface fuel treatment was necessary in dry forests to alter potential fire behaviour from crown fire to surface fire under severe fire weather conditions. This level of treatment was predicted to maintain surface fire behaviour for 30–40 years, depending on rate of understory growth, after which additional fuel treatment would be needed. We used 40% basal area removal for the dry ecoregions (maximum 60 cm diameter at breast height (DBH)), for LandFire map areas identified as having a mean

fire return interval of less than 40 years and precipitation < 650 mm.

We analysed data for changes in NPP by biomass, forest type, ecoregion, site quality and climatic conditions. We used the ecoregion-specific relationships between NPP and biomass on the previously thinned inventory and experimental plots to predict post-treatment NPP. In the wetter ecoregions (Coast Range and West Cascades), NPP tends to increase linearly at first with increasing biomass then reach a maximum where it remains fairly constant (Figure 2). In some of the drier ecoregions, NPP does not appear to reach a maximum and continues to increase. For this reason, linear regressions were only fit to plots that had not reached maximum NPP. After the plots were virtually thinned or clear-cut, the new biomass was used to predict a new NPP using the ecoregion-specific equations, producing the following year's new biomass. This was repeated until the new NPP equalled the plot NPP before treatment (law of constant final yield). For comparisons with current practices, we assumed that current landscape level NPP was constant and incremented the current plot biomass for each year using the observed NPP. The change in NPP due to external factors (e.g. N deposition) is the same for the three treatments, and we are reporting differences among treatments, so this assumption should affect the treatments equally. The total NPP resulting from the treatments were compared with total values for current management practices after 20 years.

Monte Carlo simulations were used to conduct an uncertainty analysis using the mean and standard deviations for NPP calculated by several approaches. Three alternative sets of allometric equations were used to estimate the uncertainty due to variation in region and/or species-specific allometry. The full suite of species-specific equations that use tree diameter (DBH) and height (preferred) were compared with a DBH-only national set, and to a grouped forest type set. Finally, the total uncertainty was combined with the uncertainty in land cover estimates (10%) using the propagation of error approach (National Research Council 2010).

Results

Current terrestrial carbon fluxes

The region was found to be highly productive and a strong carbon sink. The current NPP for CA, OR and WA was

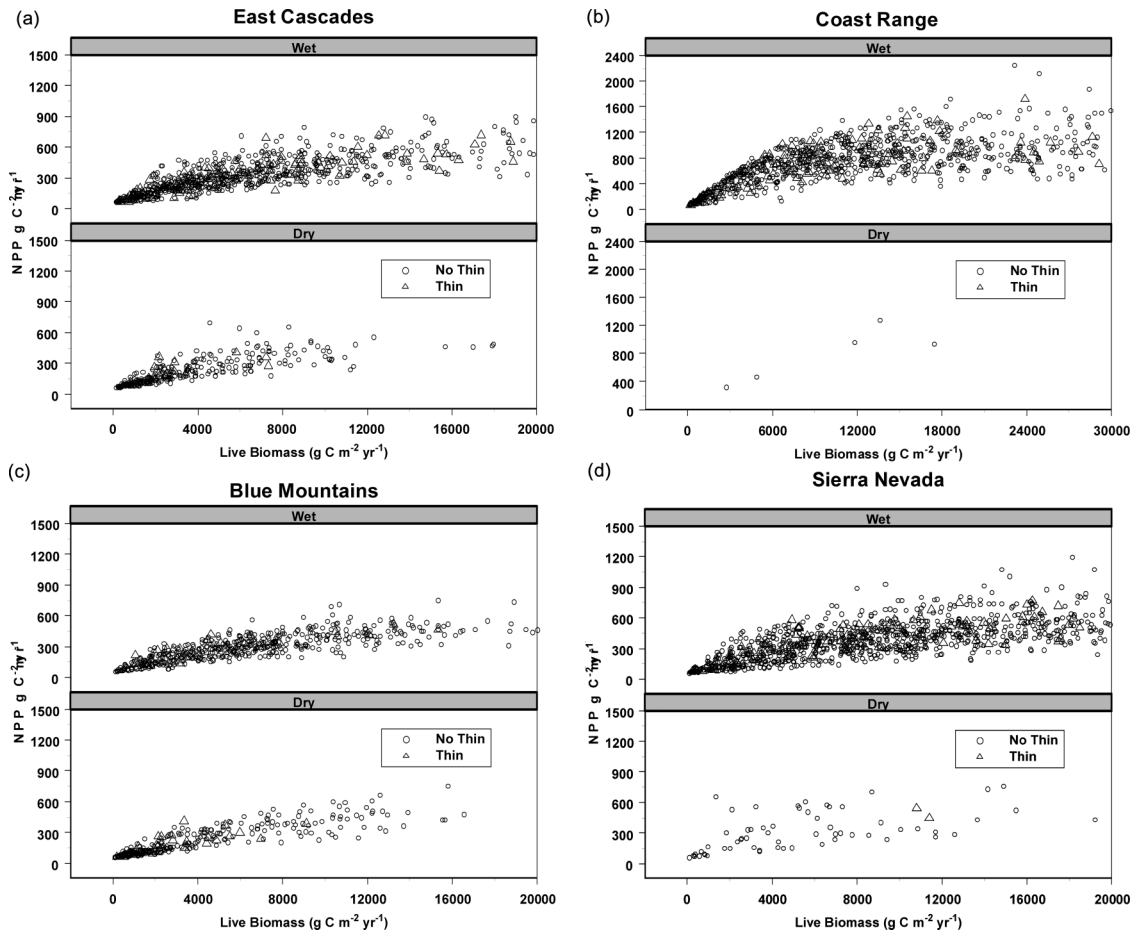

Figure 2. Relationship between NPP and biomass using inventory plot data for selected mesic (precipitation >650 mm year^{-1}) and dry (<650 mm year^{-1}) plots in selected ecoregions. (a) East Cascades (630 mm annual precipitation, all three states); (b) Coast Range (1741 mm precipitation, all three states); (c) Blue Mountains (511 mm precipitation, Oregon, Washington); (d) Sierra Nevada (915 mm precipitation, California). Data are for forests <80 years of age. Open symbols are thinned stands.

Table 2. State total and mean carbon fluxes resulting from current practices (Current NPP, NEP, fire emissions and harvest removals) and the three treatment treatments: (1) clear-cut harvest young forests (on a 50-year cycle, converting a portion of the landscape to short rotation) and setting aside mature and old forests (Sparing) in all ecoregions; (2) thin all ecoregions at moderate level (Sharing); (3) spare mesic mature and old, clear-cut mesic young and thin all dry forests (<650 mm annual precipitation) to reduce fire and drought stress in the latter (Sparing/Sharing).

State forested land (million ha)	Washington (9.0)	Oregon (12.2)	California (12.8)
Annual fossil fuel emissions (Tg C year^{-1})	21	15	105
Carbon density (Mg C ha^{-1})	172 ± 25	150 ± 22	130 ± 18
Current net primary production -NPP			
Total (Tg C year^{-1})	44.9 ± 5.2	58.3 ± 6.5	59.9 ± 6.7
Per unit area (g C m^{-2} year^{-1})	517 ± 58	477 ± 53	442 ± 50
Current net ecosystem production -NEP			
Total (Tg C year^{-1})	11.3 ± 1.2	15.2 ± 1.6	18.1 ± 2.1
Per unit area (g C m^{-2} year^{-1})	125 ± 13	125 ± 13	142 ± 16
Current fire emissions[1] (Tg C year^{-1})	0.9 ± 0.1	1.3 ± 0.2	1.8 ± 0.3
Current harvest removals (Tg C year^{-1})	5.5 ± 0.4	6.4 ± 0.5	2.7 ± 0.2
Total treatment removals (Tg C year^{-1})			
• Thin all (Sharing)	12.0 ± 0.8	14.5 ± 1.0	11.2 ± 0.8
• Cut young, spare old (Sparing)	8.0 ± 0.6	11.0 ± 0.8	8.9 ± 0.6
• Sparing in mesic / Sharing in dry	9.2 ± 0.6	10.0 ± 0.7	7.8 ± 0.6
Treatment NPP (Tg C year^{-1})			
• Thin all (Sharing)	41.2 ± 4.6	53.3 ± 6.0	54.4 ± 6.1
• Cut young, spare old (Sparing)	28.5 ± 3.2	36.4 ± 4.1	41.9 ± 4.7
• Sparing in mesic / Sharing in dry	30.9 ± 3.5	40.7 ± 4.6	44.3 ± 4.9
Area treated (million hectares)			
• Thin all (Sharing)	7.1	9.8	7.8
• Cut young, spare old (Sparing)	1.4	2.1	2.0
• Sparing in mesic / Sharing in dry	3.2	5.4	3.1

[1] Fire emissions refers to carbon loss due to direct combustion.

59.9 ± 6.7, 58.3 ± 6.5 and 44.9 ± 5.2 Tg C year^{-1}, totalling 163.0 Tg C year^{-1} and averaging about 479 g C m^{-2} year^{-1} (Table 2; Figure 3a), based on pixel values (Methods section). NEP for CA, OR and WA was 18.1 ± 2.1, 15.2 ± 1.6 and 11.3 ± 1.2 Tg C year^{-1}, respectively, averaging 125 to 142 g C m^{-2} year^{-1} (Table 2; Figure 3b). NPP was highest in the western coastal portion of the region, which is more mesic with mild temperatures due to the influence of the Pacific Ocean. Current harvest removals were significantly lower in CA (2.7 ± 0.2 Tg C year^{-1}), compared with 5.5 ± 0.4 and 6.4 ± 0.5 Tg C year^{-1} in WA and OR, respectively.

Large-scale analysis of the relationship between current NPP and biomass for inventory plots in several of the ecoregions (Figure 2) showed that the relationship between NPP and biomass was significant in all ecoregions (P values < 0.05). The range of observed values in NPP for a given biomass was quite large in some ecoregions, and a substantial portion of this variability could be attributed to spatial variation in precipitation, site quality, species composition and a heterogeneous age structure. For the majority of the ecoregions, the inclusion of forest type, precipitation and/or ownership significantly improved the relationship (P values <0.05; Table 1).

Observed thinning effects

We wanted to examine the changes in stand-level growth after thinning with our data from two thinning experiments in the region (Metolius 13 stands of which five were thinned, and Forest Hill 14 stands of which 11 were thinned).

Thinning of semi-arid ponderosa pine stands in the Metolius area (annual precipitation 360 mm) showed an increase in the ratio of above-ground NPP (ANPP) to biomass several years later compared with unthinned stands (Figure 4a). A thinning experiment in a 44-year-old ponderosa pine plantation in the Sierra Nevada Mountains, where precipitation is almost three times higher (950 mm) than the Metolius area, did not appear to alter the relationship between NPP and biomass 3 and 16 years after thinning compared with unthinned stands (Figure 4b).

Projected large-scale thinning effects

Harvest removals were highest in the Sharing treatment in all three states compared with the other treatments (Table 2), but were lowest on a per unit treated area basis. In the Sparing/Sharing treatment, the amount of biomass removed from young mesic forests (<50 years) that were clear-cut was much higher than that removed by thinning all drier forests (Figure 5), resulting in 97% and 38% removal of above-ground tree biomass per plot, respectively, which is typical of harvest practices in the region (Skog et al. 2008; Evans and Finkral 2009; Harrod et al. 2009). Removals from thinning the drier ecoregions were highest in the California Central Basin (CB) and California Chaparral and Oak Woodlands (CO) (Figure 5). Biomass removals from clear-cutting young mesic ecoregions were highest in the Washington and Oregon Coast Range (CR),

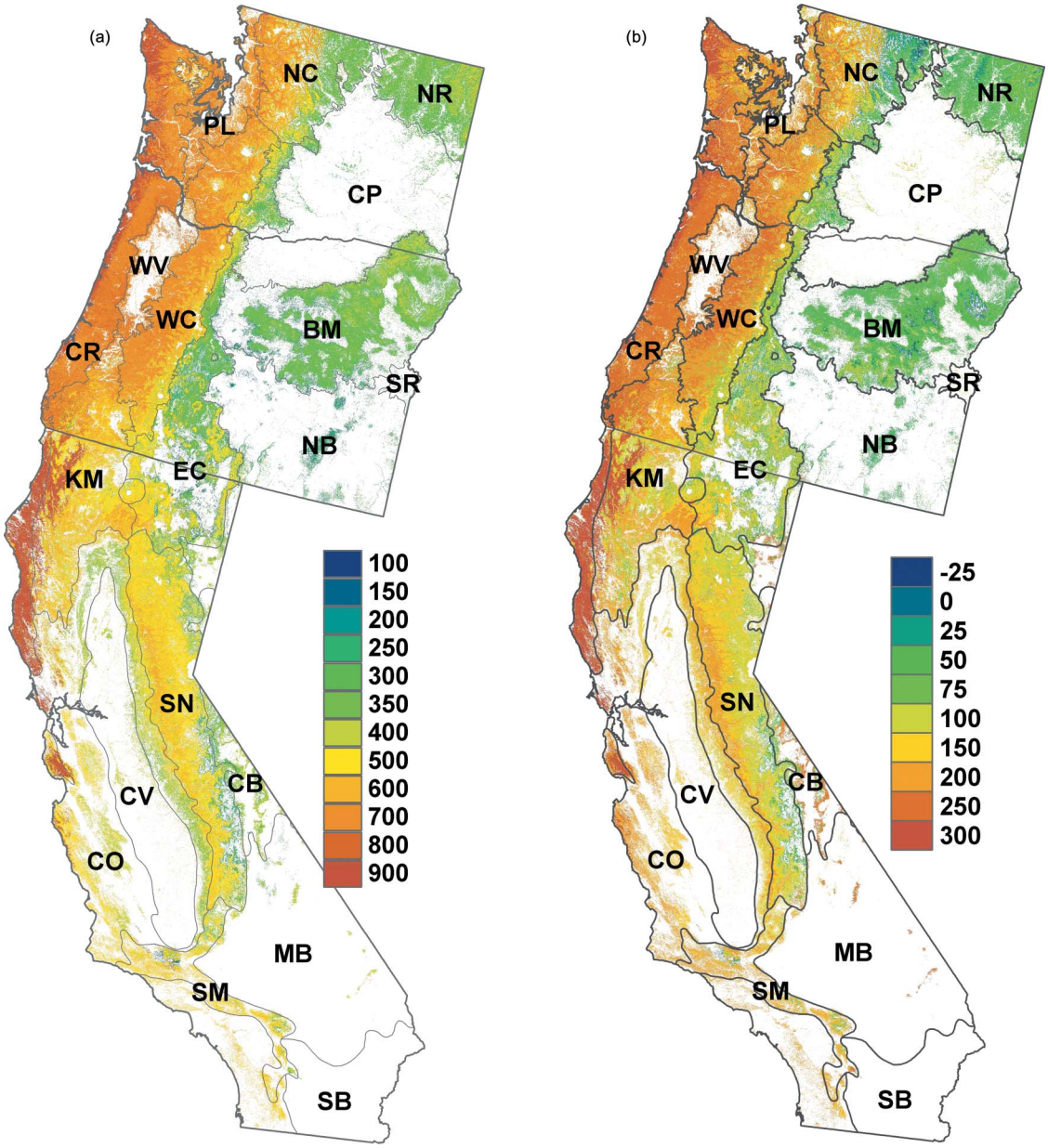

Figure 3. Regional (a) NPP and (b) NEP under current practices (CP) expressed in g C m^{-2} $year^{-1}$. BM, Blue Mountains; CB, Central Basin; CO, California Chaparral and Oak Woodlands; CP, Columbia Plateau; CR, Coast Range; CV, Central California Valley; EC, East Cascades; KM, Klamath Mountains; MB, Mohave Basin; NB, North Basin and Range; NC, North Cascades; NR, Northern Rockies; PL, Puget Lowlands; SB, Sonoran Basin; SM, Southern California Mountains; SN, Sierra Nevada; SR, Snake River; WC, West Cascades; WV, Willamette Valley.

Oregon West Cascades (WC), Oregon Willamette Valley (WV) and Washington Puget Lowland (PL) (Figure 5).

A Sharing treatment (moderately thin all forests) reduced regional NPP by 9% to 149 Tg C $year^{-1}$ after 20 years. The Sparing treatment (clear-cut all young forests, spare all mature and old forests) reduced regional NPP by 35% to 107 Tg C $year^{-1}$. In the Sparing/Sharing treatment, where land sparing was applied in the mesic mature/old forests, mesic young forests were clear-cut and thinning was applied in the dry forests as a means of reducing fire and drought stress, regional NPP was reduced by 29% to 116 Tg C $year^{-1}$. The largest decreases in NPP due to treatment were in California (Table 2), and specifically the Sierra Nevada, Southern California Mountains and California Oak Woodlands ecoregions (Figure 6), where biomass removals were highest. Other areas of high removals and large decreases in NPP were the West Cascades and North Cascades, where more area in young forests and high productivity per unit area (due to mild climate) in these ecoregions led to large decreases in NPP compared with the other ecoregions.

Mitigating the effects of drought and fire

To mitigate the effects of drought, preventing a reduction in productivity by implementing a large-scale thinning of drier

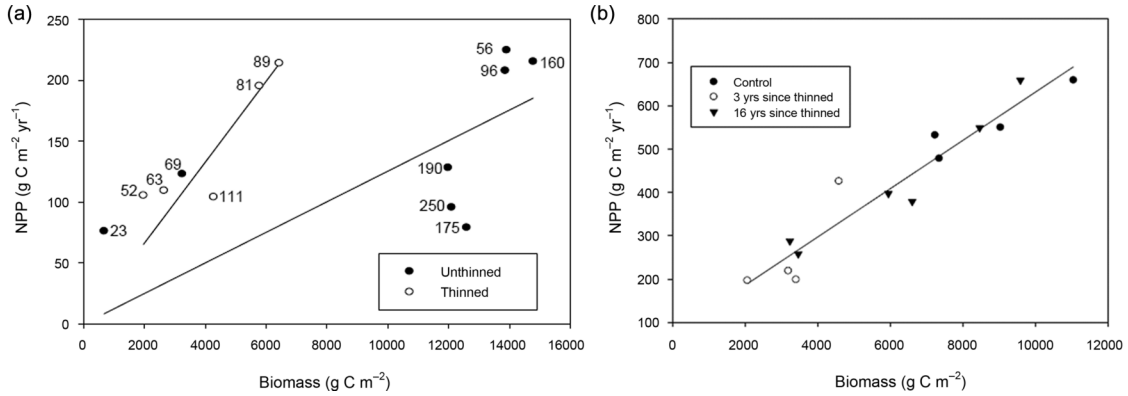

Figure 4. (a) Thinning effects on the relationship between above-ground net primary production (ANPP in g C m^{-2} year^{-1}) and biomass (g C m^{-2}) in Metolius semi-arid ponderosa pine (360 mm precipitation per year). Regressions are forced through the origin (young and mature plots $r^2 = 0.56$, old plots $r^2 = 0.27$ and overall $r^2 = 0.35$). Points are labelled with the plot mean stand age (years); (b) NPP and biomass 3–16 years post thinning compared with unthinned stands of mature (ca. 35 year) ponderosa pine in the Sierra Nevada Mountains, where annual precipitation is higher at 950 mm year^{-1}. Regressions are lines of best fit forced through the origin.

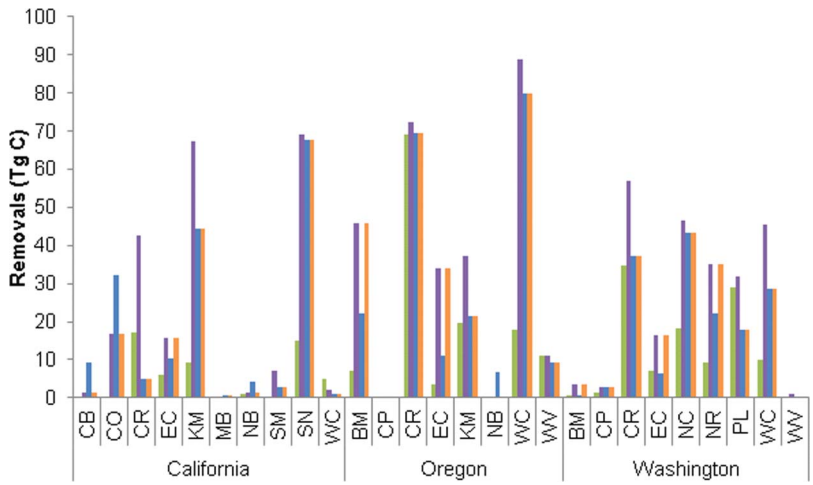

Figure 5. Comparison of current biomass removals (Tg C) in each ecoregion with (1) removals 20 years after thinning all forests (Share); (2) clear-cut young mesic forests and spare all old forests (Spare); and (3) clear-cut mesic young forests, spare mesic old forests and thin all dry forests (Spare mesic / Share Dry).

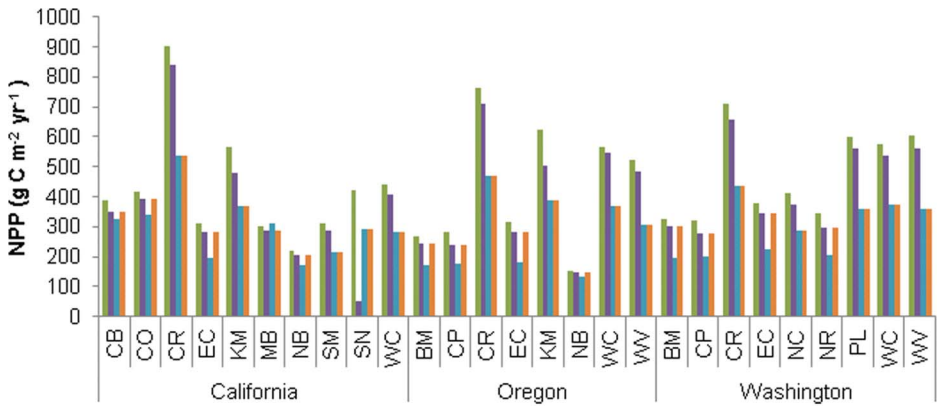

Figure 6. Comparison of current NPP (g C m^{-2} year^{-1}) in each ecoregion with (1) NPP 20 years after thinning all forests (Share); (2) clear-cut young mesic forests and spare all old forests (Spare); and (3) clear-cut mesic young forests, spare mesic old forests and thin all dry forests (Spare Mesic / Share Dry).

forests (Sparing/Sharing treatment) had a cost of a 12.5% decrease in NPP (from 41 to 36 Tg C year^{-1}).

The dry ecoregions together have a mean fire return interval of 25–55 years and 1.4 of the 12.4 million ha of dry forest land (11%) are affected by wildfires annually. As a zero-order estimate, this fire-induced loss in productive forest land corresponds to an 11% loss of NPP. Large-scale prevention of forest fires had a cost of 12.5% reduction in NPP over the 20-year treatment period (Sparing/Sharing treatment for dry forest only).

Forest fires are less frequent in the mesic regions (0.9 million ha and <5% of the forested area), but nevertheless preventive thinning has also been proposed in these areas. A Sharing treatment in which both mesic and dry forest were thinned to prevent productivity losses from wild fires results in a 9% reduction of the productivity across all ecoregions that is not offset by productivity loss due to fire under current conditions. This could change if water availability in summer decreases due to warm spring snowmelt water loss that could otherwise be available for growth in summer.

Biodiversity

In the Sparing/Sharing treatment, the amount of area of forests >50 years old that was preserved for biodiversity and adaptive capacity in the mesic forests was 18.3 million ha with 29% reduction in NPP. In the Sparing treatment, which emphasises sparing all forests >50 years old across the region, an additional 10.2 million ha are preserved for a total of 28.5 million ha.

Discussion

Current terrestrial carbon fluxes

In the Pacific Northwest US region, there is a higher frequency of younger stands on private land than on public land and there are more old stands on public land. Mean stand age on private land ranges from 42 years in the Coast Range to ca. 105 years in the California Oak Woodlands and Chaparral. Mean ages on public land range from 60 years in the Willamette Valley to ca. 130 years in the West Cascades, where most of the land is public (Hudiburg et al. 2009).

Our earlier work showed that forests of the west coast region have high biomass and indicated that regional carbon stocks could theoretically increase by 46% if forests were managed for maximum carbon storage (Hudiburg et al. 2009). Mean NPP of 80 forest types in the region was estimated at 100–900 g C m^{-2} year^{-1}, within the global range of temperate and boreal forests (100–1600 g C m^{-2} year^{-1}; Luyssaert et al. 2007).

In the mesic ecoregions, recent changes in age-class distributions were the result of implementation of the Northwest Forest Plan (NWFP) in 1993 to conserve species, such as the northern spotted owl (*Strix occidentalis caurina*) that had been put at risk from extensive harvest of older forests. Simulations showed that the area

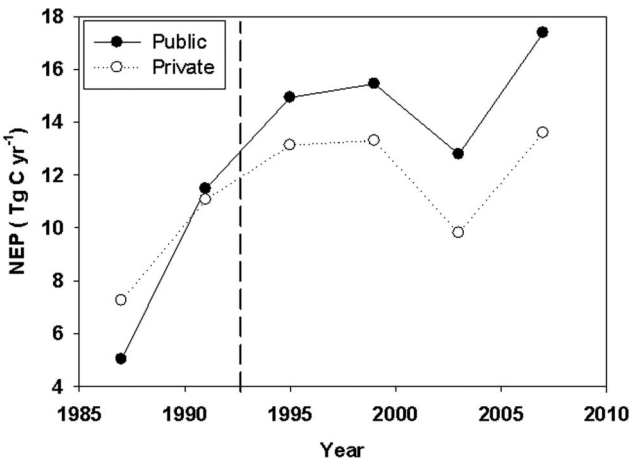

Figure 7. NEP expressed in g C m^{-2} year^{-1} after implementation of the Northwest Forest Plan on public lands in 1993 (after Turner et al. 2011). Data reported separately for public (black circles) and private (white circles) land.

of the NWFP was a carbon source (both public and private ownership) prior to implementation of the NWFP. After implementation, harvest removals were reduced by 82% on public lands and the forests became a carbon sink (Turner et al. 2011). Results from the simulations showed that NEP on public lands prior to implementation averaged 20% lower than that of private lands, and after implementation, NEP on public lands averaged 20% higher than that on private lands (Figure 7). The simulations also suggested that the drought in 2001–2003 had a large impact on NEP in all ecoregions.

Observed thinning effects

Studies have shown that thinning forests initially reduces area-based NPP while increasing NPP of the remaining trees (Law et al. 1992). Over time, regrowth after thinning results in little difference in area-based NPP compared with unthinned stands, all else being equal (Kira et al. 1953).

In the Metolius thinning analysis, the large variation in NPP for high biomass in unthinned stands was likely due to edaphic differences where the lower ANPP values at high biomass are for old forests, including a low-density stand of all old trees. The old, tall semi-arid forests survived many severe droughts and fires, as indicated by the historic fire return interval of 3–38 years in the area (Bork 1985). The thinning study on the same species but wetter climate of northern California showed no change in the relation between NPP and biomass compared with unthinned stands, indicating more rapid recovery from thinning and compensatory effects by understory vegetation, confirming the temporary effect of thinning. Although the two thinning studies were not directly comparable, they support the assumption that thinning can temporarily reduce competition in areas with more severe soil water deficits.

In the large-scale analysis of inventory plots, the inclusion of forest type, precipitation and/or ownership significantly improved the relationship between NPP and

biomass. The relationship with precipitation reflects the large regional variation in water availability. While measures of site quality (i.e. site index defined as species-specific tree height at a reference age) were available for the plot data, adequate spatially explicit regional data layers of height and age were not available for scaling plot data. In a previous analysis with inventory data (Hudiburg et al. 2009), we found a significant difference between the site indexes of publically and privately owned forests due to the historical pattern of private land ownership being located in lower elevation forested areas characterised by higher productivity. When we accounted for ownership (as a surrogate for site quality), the NPP-biomass relationship improved in about half of the ecoregions (Table 1). Thus, we concluded that, in general, over the treatment period of 20 years, there is no long-term thinning response of NPP (i.e. it does not exceed that of unthinned stands).

Projected large-scale thinning effects

The Sharing treatment resulted in the smallest decline in regional NPP after 20 years, and it also had the highest total harvest removals (Table 2). However, on a per-unit-treated-area basis, the Sparing treatment (clear-cutting young forests <50 years) removed 50 times the biomass of the Sharing treatment (thinning). The higher total removals in the Sharing treatment were somewhat offset by more rapid recovery of NPP. This result is due to a combination of much lower initial decline in NPP on thinned versus clear-cut plots and differences in treated area. The Coast Range in Oregon is a good example of this because the total removals are nearly equal for each treatment (Figure 5), but the effect on NPP was quite different (Figure 6). Thinning the same total amount of biomass over a much larger area does not reduce overall NPP as much as clearing all young forests over a smaller area. For the region in general, the larger initial loss in NPP from clearing young forests takes much longer to recover because of the initiation stage of growth.

The large decreases in NPP of the Sparing/Sharing treatment were influenced by historical harvest practices. Harvest had reached a peak on both public and private lands prior to implementation of the NWFP in the early 1990s, which means there were many young forests in some ecoregions, like the historically productive Coast Range and West Cascades, which were subject to clear-cuts in this treatment.

All the proposed large-scale changes in forest management were expected to result in a considerable decrease of NPP over a 20-year time period. Whether the NPP reduction would be reflected in the regional carbon sink (net biome production) ultimately depends on the C-losses through harvest, fire and decomposition.

Mitigating the effects of drought and fire

Our results (Figure 4a, b) support the assumption that thinning temporarily reduces competition in areas with more severe soil water deficits. As a consequence, large-scale thinning of semi-arid regions could mitigate the effects of summer droughts on forest production. During the 2001–2003 extreme droughts, the Metolius semi-arid mature forest experienced a 40–44% decrease in gross photosynthesis and NEP and a ca. 15% decrease in NPP compared with surrounding years (Thomas et al. 2009). Preventing a reduction in productivity by implementing a large-scale thinning of drier forests (Sparing/Sharing treatment) still led to a 12.5% decrease in NPP. However, if thinning could avoid large-scale dieback of forests due to drought stress, preventive thinning may have a role to play under future climate conditions if a substantial increase in drought stress and frequency is predicted for the dry ecoregions.

The zero-order estimate of fire-induced loss in NPP (11%) was less than that of large-scale prevention of forest fires (12.5%) over the 20-year treatment period (Sparing/Sharing treatment for dry forest only). Currently, high-severity fires responsible for high basal area mortality account for only about 20% of burned areas in the region (Schwind 2008; Meigs et al. 2009). In the Metolius area, NPP several years after fire was only 40% lower in the high versus low-severity burn areas, suggesting compensatory effects of new vegetation growth. Reduced NEP was primarily due to change in NPP, not heterotrophic respiration (Meigs et al. 2009). This and other studies suggest that the net effect of thinning the drier forests would be reduced NPP compared with NPP after fires.

Biodiversity

Treatment effects on biodiversity could not be determined. However, previous studies in the region showed that thinning encouraged growth of important mid-canopy layers of plant species in structurally complex mesic forests (Comfort et al. 2010), and in Sierra mixed conifer forests higher plant species richness was associated with less canopy closure (a measure of thinning intensity; Battles et al. 2001). In addition, light-to-moderate thinning had a neutral-to-positive influence on bird species diversity in the Pacific Northwest (Hayes et al. 1997), and this was likely due to increased shrub and understory layers and structural diversity. A meta-analysis across North America concluded that the magnitude of response to forest thinning is often small for several years after thinning; however, some species of higher conservation concern may be positively or negatively affected by thinning and simple diversity and richness measures may not be sufficient for fully understanding the effects of thinning on biodiversity (Verschuyl et al. 2011).

Current harvest practices treat approximately 1.1% of the total forested area annually and remove a total of 14.6 ± 1.0 Tg C year^{-1} from all three states combined, with 44% of this harvest from Oregon alone (Smith et al. 2007). While all of the treatments included considerable harvest increases compared with current management practices, the proposed harvest area ranged from < 1–3.6% of the total forest area annually and did not exceed historical harvest rates (Table 2).

In the Sparing/Sharing treatment, the area of mesic young forests treated with clear-cut harvest (5%/year harvest of area designated for the treatment) was still within historical rates when scaled to the total forest area. In this treatment, the area spared in mesic forests tripled the land area preserved compared with currently preserved land area (4.6 million ha in the region), which is small relative to that in other temperate regions of the world. In the Sparing treatment, the land area preserved increased to 28.5 million ha, over six times current levels; however, the cost in reduced NPP was the highest of all treatments (35% vs. 29% for Sparing/Sharing). Therefore, if the Sparing/Sharing treatment was selected as the best approach for sustaining the region's biodiversity and supporting adaptation and migration through functional connectivity of forest, while reducing drought stress in dry forests by thinning, the cost in terms of productivity would be high (29% reduction in NPP). However, under the current environmental conditions this lower NPP would still sustain a substantial carbon sink (Hudiburg et al 2009; Luyssaert et al 2008).

Conclusions

The regional analysis indicates that proposed climate change mitigation actions to reduce impacts in Pacific Northwest US forests that included a treatment of sparing mesic mature forests, clear-cutting mesic young forests (<50 years, reducing harvest cycle from 80 to 50 years, which is already being planned) and thinning all age classes of dry forests to minimise drought and fire impacts on carbon (<650 mm precipitation per year) resulted in a 29% decrease in NPP over the 20-year treatment period compared with NPP resulting from current practices. Emphasising sparing of all old forests, mesic and dry, while clear-cutting all young forests results in the largest reduction in NPP (35%). The impact could be reduced by treating a subset of the young forests, which may also be desirable for facilitating migration of trees to a more favourable climate, and allow recruitment of young into older age classes or acceleration of old-growth structure. Thinning of all forests at a moderate level would have the lowest impact on NPP (9% reduction), but it would not preserve mature and old forests. It could also reduce occurrence of spatial complexity in early successional forests that is similar to that in old-growth forests (Donato et al. 2011). There are trade-offs with each treatment, but this study indicates they come at a cost of reducing regional NPP over 20 years in this region. In particular, thinning or managing for fire suppression may remove more NPP with a longer time-lag for recovery than fire itself. Repeated cycles of thinning in dry forests (20–30-year cycle) and clear-cutting young forests (50-year cycle) would likely lead to less of a reduction in regional NPP over the following 50-year period because NPP of some preserved forests may decrease due to ageing and NPP of the thinned dry forests may not be reduced much further after the initial harvest and regrowth.

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Author contributions

B.L designed the study, and conducted and guided field studies and analysis. T.H. conducted the regional analysis and statistical analysis on plot data. S.L. contributed to the analysis. B.L., T.H. and S.L. co-wrote the paper.

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