To: Oregon Board of Forestry

From: Bob Van Dyk, Wild Salmon Center

Re: State Forests (Agenda Item 9) at the July 24 Board of Forestry meeting

Date: July 23, 2018

Dear Board Members,

This document conveys the views of the Wild Salmon Center regarding Agenda Item 9 at the upcoming July 24 Board of Forestry (BOF) Meeting. We have consulted with a variety of partners on these comments, but due to time limitations this document is our own.

We recommend that the BOF table action on Item 9; more work is needed before moving forward on Forest Management Plan (FMP) revisions. We recommend instead a more intensive effort to advance a Habitat Conservation Plan (HCP), as well as a short-term focus on clarifying future costs and revenues of the state forest program.

I: Planning Context:

In June of 2013, five years ago, the Board of Forestry directed staff to work toward a new FMP that would improve financial viability and improve conservation outcomes. As we stated at the time, we saw little evidence that such a "win-win" approach existed -- if increased harvests were the path to financial viability. We stated that revenue diversification was the more likely path to the twin goals.

At the time of this direction for a new FMP, the primary driver in the conversation seemed to be the balance of the Forest Development Fund (FDF), which pays the state forest's bills. Long-term projections noted a declining FDF balance, which was interpreted as a threat to financial viability. No analysis or goal related to conservation outcomes was described. No clear or operationalized definitions of financial viability nor conservation were provided.

This 2013 Board direction to explore a new FMP coincided with the abandonment of a management framework designed to address just these sorts of questions. Because of the inherent vagueness of concepts like "conservation" and "financial viability," an earlier board spent several years developing a set of performance measures to track these very issues.[1] Targets were set and tracked over time, in the context of adaptive management. Annual reports on these performance measures were completed. These well-conceived and measureable outcomes were quietly abandoned.

A stakeholder group on a new FMP was assembled to consider alternatives. Stakeholder proposals were solicited and shared for scientific review. Not surprisingly, the conservation alternatives were found to increase rare habitats but reduce timber harvest, while timber proposals (such as the 70-30 plan) were found to generate more revenue but reduce conservation values. No "win-win" of increased harvest and increased conservation was identified.

Despite the negative conservation findings of the science review regarding the 70-30 approach, the Board directed staff to further explore a 70-30 strategy, which had been advocated by Hampton Lumber, a privately-held industrial timber company who buys many state timber sales. WSC provided testimony at

[1] For example, on financial viability the Board developed a measure of the cost to "fully implement the FMP." On conservation, indicators were developed for water quality and for quantity of habitat.

the time expressing opposition to this approach for two reasons: 1) The 70-30 approach had just been evaluated by a science panel and found to have many negative outcomes for conservation. 2) Though conservation was a purported goal, none of the Board members asked for nor specified conservation strategies that would be improved by a 70-30 approach.

More intensive modeling of the 70-30 approach provided no evidence of positive conservation outcomes, nor of long-term financial viability, but it did clarify that the department needed to revisit the forest inventory. Another intensive process was undertaken, including the assembly of a technical group (TERG). A report was provided, and the underlying inventory improved.

II: Need for Analysis of Current and Future Financial Situation

After five years of process, there is still no clear analysis of the long-term prospects for revenues and costs for the state forest program. In particular, we think an analysis is needed of the likely future costs of the state forest program. As we presented to the Board twice this year, ODF's costs have rapidly outstripped inflation since the FMP planning process began. If financial viability, defined as resources adequate to fund FMP implementation, is to be attained, there must be a realistic analysis of current and future costs anticipated by the ODF state forest program. Without a sense of the range of future costs of the state forest program, it is not possible to gauge the kind of changes that might be needed to the FMP.

III: The Staff Report and Draft Guiding Principles:

A) General Comment: The draft guiding principles need clearer presentation and explanation. As you know, the current FMP has 14 guiding principles. The revised principles (10 in April, now 11) build on much of the language of the current 14 principles, but that is unclear from the staff report and staff proposal. Instead, the new guiding principles are presented without reference to the existing FMP language. The differences between the current principles and the proposed principles are not explained. A superior approach would have been to use line edits of the current principles (given the many similarities), and to include explanations for any changes. For explanations, it would be helpful to know how proposed changes reflected efforts to attain the twin goals of financial viability and improved conservation outcomes, or to respond to other developments, such as the need to clarify a principle, or the relevance of more recent science. Such an approach would have allowed the reader to understand not only the "what" of changes to the current principles as well as the "why." As it is, the reader is left to guess at the purpose of the changes and how they relate to the goals of the revision.[2]

The principles would have also benefitted from short labels: e.g. GPV, Conservation, Financial Viability.

B) Definitions of Financial Viability and Conservation: The definitions offered for Conservation and Financial Viability provided in the staff report are unobjectionable on their faces, but they are also not very helpful to decision making. The definitions offer broad language and little specific sense of what the terms mean for prioritization of a new state forest plan. As we noted above, the BOF already has approved performance measures for conservation and financial viability that might prove useful in thinking specifically about the these values.

C) Indicators: The staff report offers an impacts analysis framework to be used to assess a potential

---

[2] We conveyed our desire for greater explanations to ODF staff, who kindly prepared and provided a table with comparisons, but we found it difficult to follow and have not had time since receiving it late last week to follow up with staff on it.

change to the FMP. The framework follows the approach taken in assessing proposals from the 2013 Stakeholder Group. The framework provides a viable method for evaluating changes to the FMP, though certainly we will be interested in learning more about the method to select and evaluate these indicators should this approach be implemented.

D) Suggested Revisions: We offer the specific suggestions and comments on the guiding principles. Proposed revisions are in **bold text**.

Principle 2: State forests will be managed, conserved, and restored to provide overall biological diversity of state forest lands **and contribute to the recovery of species of concern**, including the variety of habitats for native fish and wildlife, and support the accompanying ecological processes. The Greatest Permanent Value and Forest Management Planning rules are the Board's expression of providing conservation.

> Explanation: Board Member Williams requested language on recovery of species of concern at the April BOF meeting. We support her suggestion as an affirmation that state forest lands have a role to play in species recovery. The character and extent of that role will, of course, follow future decisions.

Principle 3: The plan will provide revenue to ensure financial viability and sustain the values that support GPV, **partly by diversifying revenue sources and improving business practices.**

> Explanation: The principle as written suggests that the forest management plan will be sufficient to provide the range of GPV goals, but historically the FMP has not considered revenues beyond timber receipts. The principle as proposed could thus be read to mean that timber harvests alone must be sufficient to meet GPV goals. Recent history makes clear that even record timber revenues do not adequately sustain GPV goals. We believe the board should make clear that revenue diversification is a core principle of a new FMP.

Principle 8: The plan will comply with state and federal laws and rules **and prioritizes a habitat conservation plan as the desired approach to ensuring GPV.**

> Explanation: Recent history on federal lands and on the Elliott State Forest make clear that federal requirements to protect endangered species are a critical component of public land management. A habitat conservation plan provides a mechanism to contribute to species recovery while also helping to ensure activities such as timber production. While the Board is pursuing an HCP, it has never made the attainment of an HCP a priority. We believe it is time to do so.

Principle 11: On climate change.

> Comment: We support the proposed language and are pleased to see the recognition of climate change adaptation and mitigation as important components of the state forest plan.

IV: Closing

Thank you for the opportunity to present these comments. We will be happy to discuss them further.

On State Forest Management Options

Board of Forestry Mtg, July 2018

Bob Van Dyk

WILD SALMON CENTER

Organization of Presentation

- Recommendation
- Planning Context
- Comments on Principles

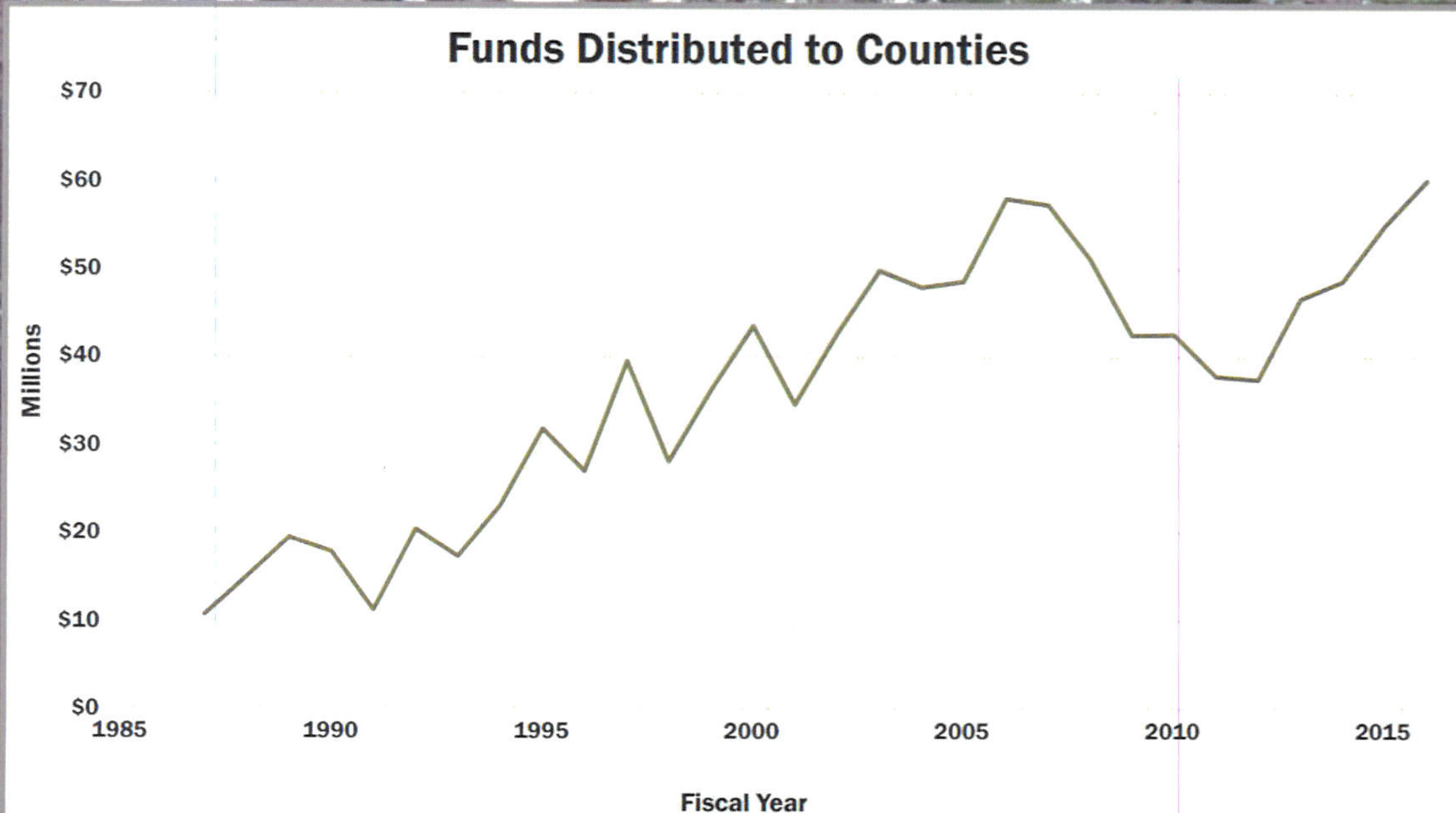
Recommendation

- Table Agenda Item
- Clarify financial viability goal
  - Future Costs
  - Future Revenues
  - Adequacy of current program, investment needs
- Clarify HCP standards
- Then proceed if appropriate…

Planning Context: Revenue and Costs

- Very high revenues
- Very high costs

County share of revenue
to counties, local taxing districts

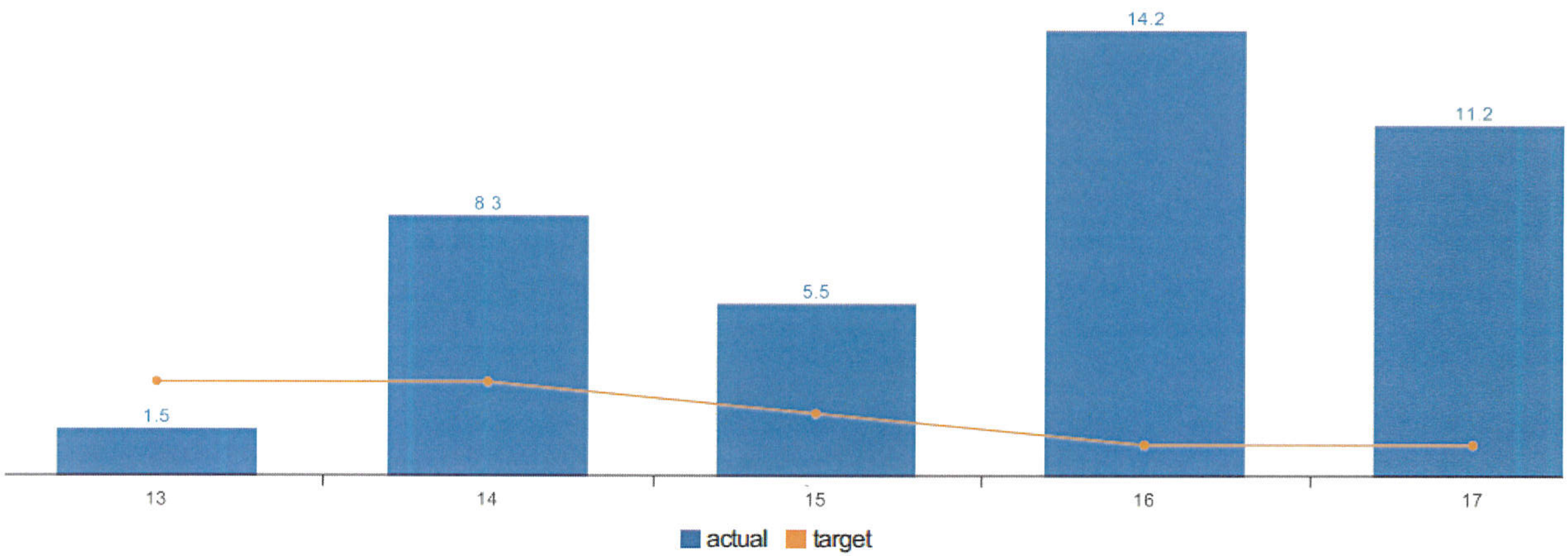


State Forests

Key Performance Measure for Revenue.

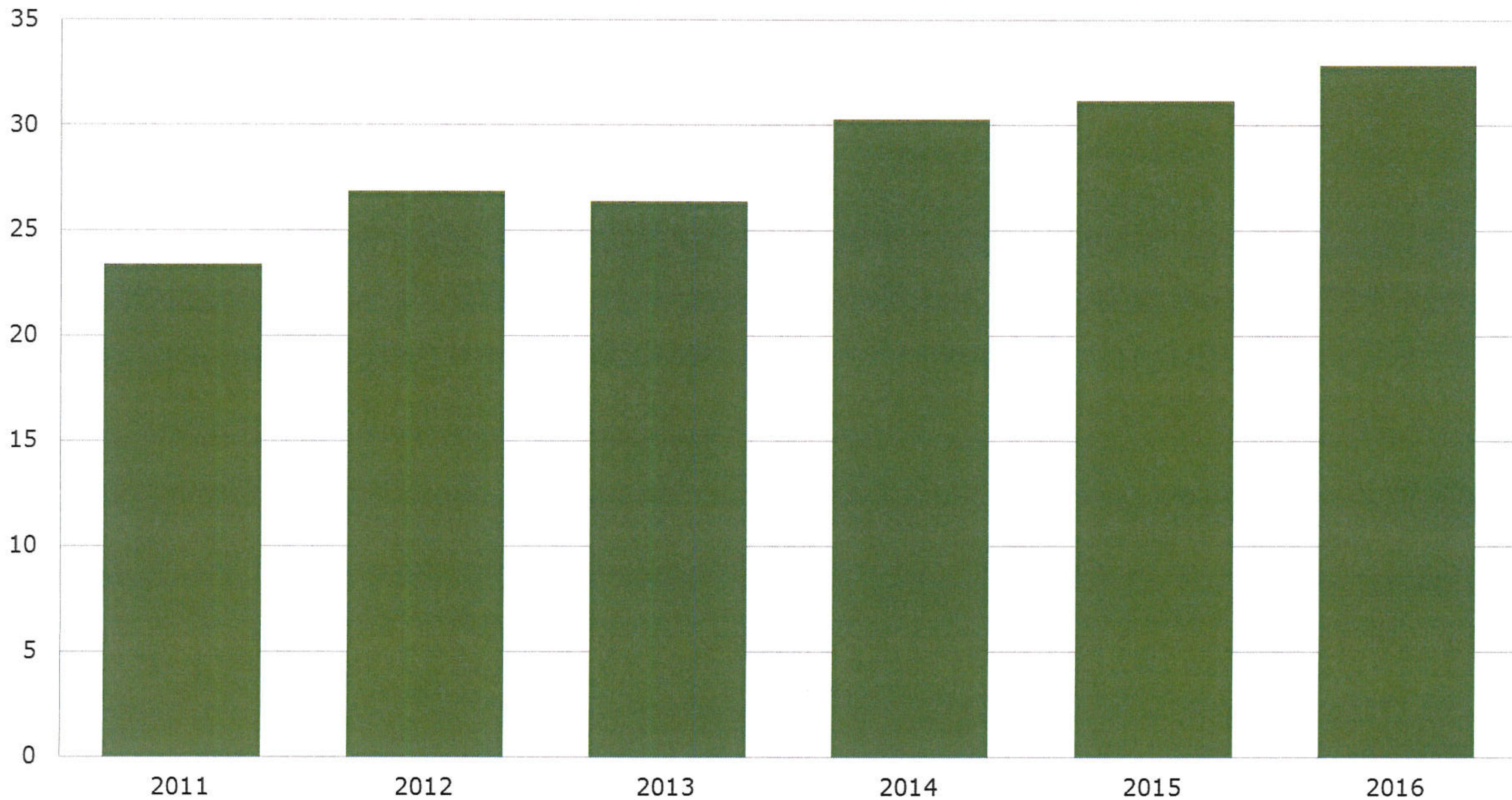
5 STATE FORESTS TOTAL REVENUE - Percent increase in total revenue produced by State Forests
Data Collection Period: Jul 01 - Jun 30

'd Trend = positive result



41% increase in six years (inflation 8%)

STATE FOREST COSTS 2011-2016 (millions of dollars)



Planning Context: Disinvestment

- Much slimmer staff (appx 270 FTE v 190 FTE)
- Targeting of high value stands "with favorable return to FDF" (Forest Development Fund)
- "Sales that represent a forest investment, don't cover all costs, or don't contribute revenue to the Division should be avoided or added… after Revenue Targets are achieved."

New Science

- Climate
- Fire vulnerability of plantations
- Streamflow

What next?

- <u>Forest Plan is not the problem</u>

- Revenues are record high
- But austerity program

- Focus on solutions outside of forest plan

Principles

- Poorly presented -- no line edits v. current plan
- Unexplained -- no narrative for changes
- Vague
- Result: No clarification of choices, direction

Specific Comments on Principles

- Add focus on revenue diversification as a strategy
- Add focus on HCP
- Add reference to contributing to species recovery

Principle 2

- State forests will be managed, conserved, and restored to provide overall biological diversity of state forest lands **and contribute to the recovery of species of concern**, including the variety of habitats for native fish and wildlife, and support the accompanying ecological processes. The Greatest Permanent Value and Forest Management Planning rules are the Board's expression of providing conservation.

Principle 3

- The plan will provide revenue to ensure financial viability and sustain the values that support GPV, **partly by diversifying revenue sources and improving business practices.**

Principle 8

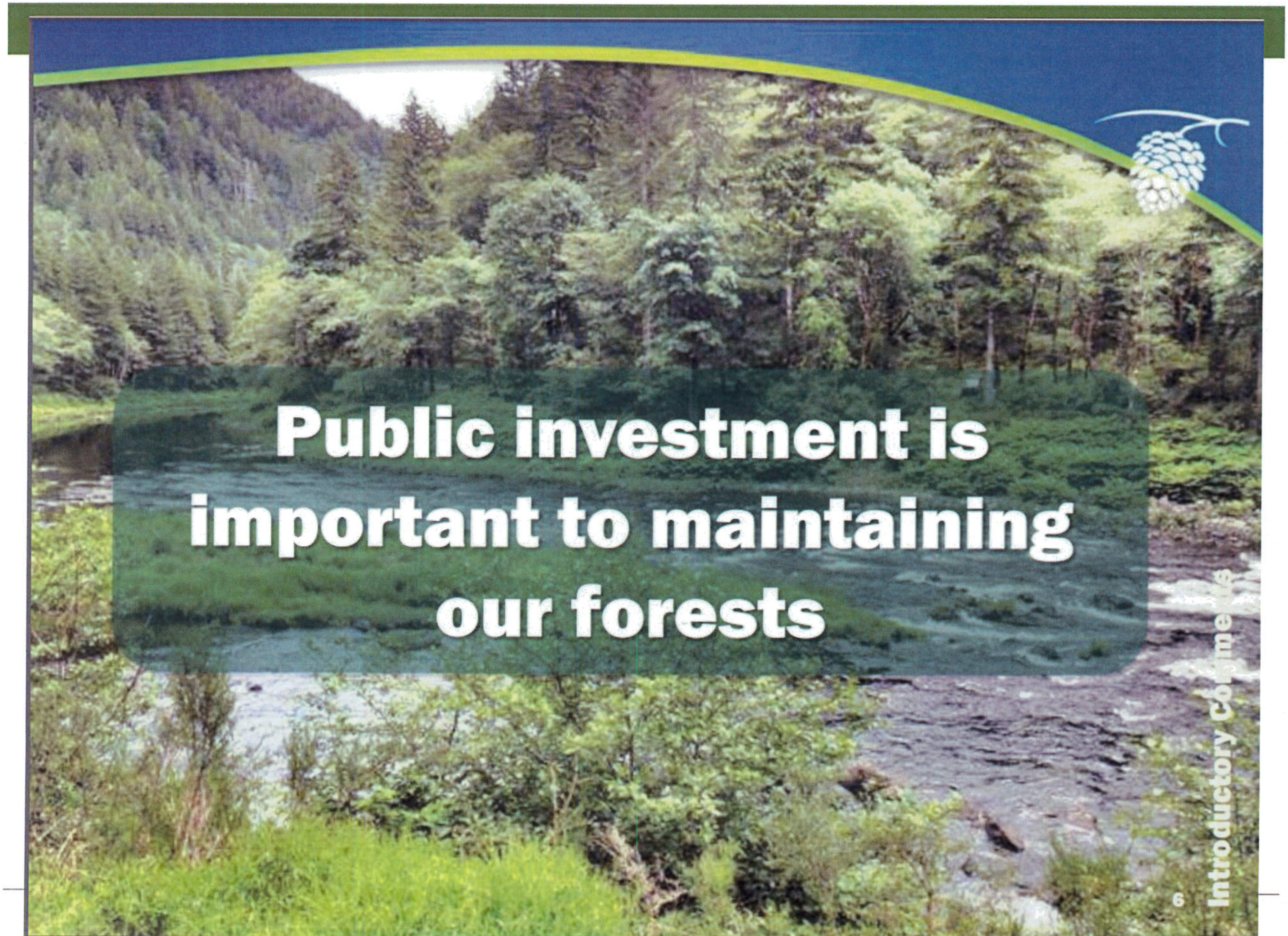
- The plan will comply with state and federal laws and rules **and prioritizes a habitat conservation plan as the desired approach to ensuring GPV.**

New Principle 11 on Climate Change

- We support.

Toxic Chemicals

- Increasing concern from public
- Especially aerial application of pesticides
- Need for specific language and/or incorporation of strategy
  - Limited Use
  - Public Transparency, including notice


Public investment is
important to maintaining
our forests
Introductory
6


The bottom line
Public investment is critical:
• To protect forests from fire
• To keep forests healthy and working
• To manage our state-owned forests for many benefits
• To help restore federal forests
Governor's Budget
32

Land use strategies to mitigate climate change in carbon dense temperate forests

Beverly E. Law[a,1], Tara W. Hudiburg[b], Logan T. Berner[c], Jeffrey J. Kent[b], Polly C. Buotte[a], and Mark E. Harmon[a]

[a]Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97333; [b]Department of Forest, Rangeland, and Fire Sciences, University of Idaho, Moscow, ID 83844; and [c]EcoSpatial Services L.L.C., Flagstaff, AZ 86004

Edited by William H. Schlesinger, Duke University, Durham, NC, and approved January 22, 2018 (received for review November 16, 2017)

Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but ecosystem process-based integration of climate change, enhanced CO_2, disturbance from fire, and management actions at regional scales are extremely limited. Here, we examine the relative merits of afforestation, reforestation, management changes, and harvest residue bioenergy use in the Pacific Northwest. This region represents some of the highest carbon density forests in the world, which can store carbon in trees for 800 y or more. Oregon's net ecosystem carbon balance (NECB) was equivalent to 72% of total emissions in 2011–2015. By 2100, simulations show increased net carbon uptake with little change in wildfires. Reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increase NECB 56% by 2100, with the latter two actions contributing the most. Resultant cobenefits included water availability and biodiversity, primarily from increased forest area, age, and species diversity. Converting 127,000 ha of irrigated grass crops to native forests could decrease irrigation demand by 233 billion $m^3 \cdot y^{-1}$. Utilizing harvest residues for bioenergy production instead of leaving them in forests to decompose increased emissions in the short-term (50 y), reducing mitigation effectiveness. Increasing forest carbon on public lands reduced emissions compared with storage in wood products because the residence time is more than twice that of wood products. Hence, temperate forests with high carbon densities and lower vulnerability to mortality have substantial potential for reducing forest sector emissions. Our analysis framework provides a template for assessments in other temperate regions.

forests | carbon balance | greenhouse gas emissions | climate mitigation

Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but regional assessments to determine feasibility, timeliness, and effectiveness are limited and rarely account for the interactive effects of future climate, atmospheric CO_2 enrichment, nitrogen deposition, disturbance from wildfires, and management actions on forest processes. We examine the net effect of all of these factors and a suite of mitigation strategies at fine resolution (4-km grid). Proven strategies immediately available to mitigate carbon emissions from forest activities include the following: (*i*) reforestation (growing forests where they recently existed), (*ii*) afforestation (growing forests where they did not recently exist), (*iii*) increasing carbon density of existing forests, and (*iv*) reducing emissions from deforestation and degradation (1). Other proposed strategies include wood bioenergy production (2–4), bioenergy combined with carbon capture and storage (BECCS), and increasing wood product use in buildings. However, examples of commercial-scale BECCS are still scarce, and sustainability of wood sources remains controversial because of forgone ecosystem carbon storage and low environmental cobenefits (5, 6). Carbon stored in buildings generally outlives its usefulness or is replaced within decades (7) rather than the centuries possible in forests, and the factors influencing product substitution have yet to be fully explored (8). Our analysis of mitigation strategies focuses on the first four strategies, as well as bioenergy production, utilizing harvest residues only and without carbon capture and storage.

The appropriateness and effectiveness of mitigation strategies within regions vary depending on the current forest sink, competition with land-use and watershed protection, and environmental conditions affecting forest sustainability and resilience. Few process-based regional studies have quantified strategies that could actually be implemented, are low-risk, and do not depend on developing technologies. Our previous studies focused on regional modeling of the effects of forest thinning on net ecosystem carbon balance (NECB) and net emissions, as well as improving modeled drought sensitivity (9, 10), while this study focuses mainly on strategies to enhance forest carbon.

Our study region is Oregon in the Pacific Northwest, where coastal and montane forests have high biomass and carbon sequestration potential. They represent coastal forests from northern California to southeast Alaska, where trees live 800 y or more and biomass can exceed that of tropical forests (11) (Fig. S1). The semiarid ecoregions consist of woodlands that experience frequent fires (12). Land-use history is a major determinant of forest carbon balance. Harvest was the dominant cause of tree mortality (2003–2012) and accounted for fivefold as much mortality as that from fire and beetles combined (13). Forest land ownership is predominantly public (64%), and 76% of the biomass harvested is on private lands.

Significance

Regional quantification of feasibility and effectiveness of forest strategies to mitigate climate change should integrate observations and mechanistic ecosystem process models with future climate, CO_2, disturbances from fire, and management. Here, we demonstrate this approach in a high biomass region, and found that reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increased net ecosystem carbon balance by 56% by 2100, with the latter two actions contributing the most. Forest sector emissions tracked with our life cycle assessment model decreased by 17%, partially meeting emissions reduction goals. Harvest residue bioenergy use did not reduce short-term emissions. Cobenefits include increased water availability and biodiversity of forest species. Our improved analysis framework can be used in other temperate regions.

Author contributions: B.E.L. and T.W.H. designed research; B.E.L., T.W.H., and P.C.B. performed research; M.E.H. contributed new reagents/analytic tools; B.E.L., L.T.B., J.J.K., and P.C.B. analyzed data; B.E.L., T.W.H., L.T.B., and M.E.H. wrote the paper; and M.E.H. contributed the substitution model.

The authors declare no conflict of interest.

This article is a PNAS Direct Submission.

Data deposition: The CLM4.5 model data are available at Oregon State University (terraweb.forestry.oregonstate.edu/FMEC). Data from the >200 intensive plots on forest carbon are available at Oak Ridge National Laboratory (https://daac.ornl.gov/NACP/guides/NACP_TERRA-PNW.html), and FIA data are available at the USDA Forest Service (https://www.fia.fs.fed.us/tools-data/).

[1]To whom correspondence should be addressed. Email: bev.law@oregonstate.edu.

This article contains supporting information online at www.pnas.org/lookup/suppl/doi:10.1073/pnas.1720064115/-/DCSupplemental.

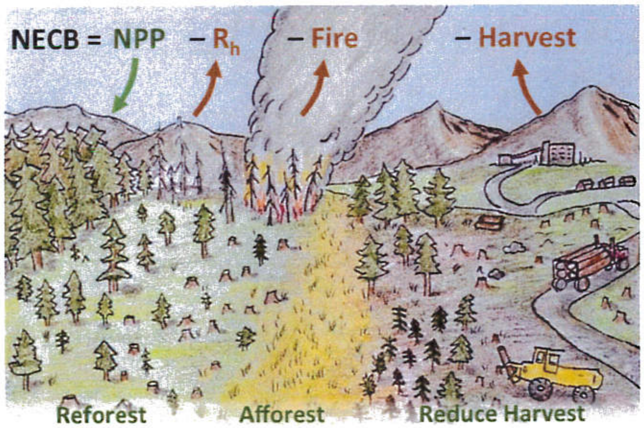


Fig. 1. Approach to assessing effects of mitigation strategies on forest carbon and forest sector emissions. NECB is productivity (NPP) minus Rh and losses from fire and harvest (red arrows). Harvest emissions include those associated with wood products and bioenergy.

Many US states, including Oregon (14), plan to reduce their greenhouse gas (GHG) emissions in accordance with the Paris Agreement. We evaluated strategies to address this question: How much carbon can the region's forests realistically remove from the atmosphere in the future, and which forest carbon strategies can reduce regional emissions by 2025, 2050, and 2100? We propose an integrated approach that combines observations with models and a life cycle assessment (LCA) to evaluate current and future effects of mitigation actions on forest carbon and forest sector emissions in temperate regions (Fig. 1). We estimated the recent carbon budget of Oregon's forests, and simulated the potential to increase the forest sink and decrease forest sector emissions under current and future climate conditions. We provide recommendations for regional assessments of mitigation strategies.

Results

Carbon stocks and fluxes are summarized for the observation cycles of 2001–2005, 2006–2010, and 2011–2015 (Table 1 and Tables S1 and S2). In 2011–2015, state-level forest carbon stocks totaled 3,036 Tg C (3 billion metric tons), with the coastal and montane ecoregions accounting for 57% of the live tree carbon (Tables S1 and S2). Net ecosystem production [NEP; net primary production (NPP) minus heterotrophic respiration (Rh)] averaged 28 teragrams carbon per year (Tg C y^{-1}) over all three periods. Fire emissions were unusually high at 8.69 million metric tons carbon dioxide equivalent (tCO_2e y^{-1}, i.e., 2.37 Tg C y^{-1}) in 2001–2005 due to the historic Biscuit Fire, but decreased to 3.56 million tCO_2e y^{-1} (0.97 Tg C y^{-1}) in 2011–2015 (Table S4). Note that 1 million tCO_2e equals 3.667 Tg C.

Our LCA showed that in 2001–2005, Oregon's net wood product emissions were 32.61 million tCO_2e (Table S3), and 3.7-fold wildfire emissions in the period that included the record fire year (15) (Fig. 2). In 2011–2015, net wood product emissions were 34.45 million tCO_2e and almost 10-fold fire emissions, mostly due to lower fire emissions. The net wood product emissions are higher than fire emissions despite carbon benefits of storage in wood products and substitution for more fossil fuel-intensive products. Hence, combining fire and net wood product emissions, the forest sector emissions averaged 40 million tCO_2e y^{-1} and accounted for about 39% of total emissions across all sectors (Fig. 2 and Table S4). NECB was calculated from NEP minus losses from fire emissions and harvest (Fig. 1). State NECB was equivalent to 60% and 70% of total emissions for 2001–2005 and 2011–2015, respectively (Fig. 2, Table 1, and Table S4). Fire emissions were only between 4% and 8% of total emissions from all sources (2011–2015 and 2001–2004, respectively). Oregon's forests play a larger role in meeting its GHG targets than US forests have in meeting the nation's targets (16, 17).

Historical disturbance regimes were simulated using stand age and disturbance history from remote sensing products. Comparisons of Community Land Model (CLM4.5) output with Forest Inventory and Analysis (FIA) aboveground tree biomass (>6,000 plots) were within 1 SD of the ecoregion means (Fig. S2). CLM4.5 estimates of cumulative burn area and emissions from 1990 to 2014 were 14% and 25% less than observed, respectively. The discrepancy was mostly due to the model missing an anomalously large fire in 2002 (Fig. S3*A*). When excluded, modeled versus observed fire emissions were in good agreement ($r^2 = 0.62$; Fig. S3*B*). A sensitivity test of a 14% underestimate of burn area did not affect our final results because predicted emissions would increase almost equally for business as usual (BAU) management and our scenarios, resulting in no proportional change in NECB. However, the ratio of harvest to fire emissions would be lower.

Projections show that under future climate, atmospheric carbon dioxide, and BAU management, an increase in net carbon uptake due to CO_2 fertilization and climate in the mesic ecoregions far outweighs losses from fire and drought in the semiarid ecoregions. There was not an increasing trend in fire. Carbon stocks increased by 2% and 7% and NEP increased by 12% and 40% by 2050 and 2100, respectively.

We evaluated emission reduction strategies in the forest sector: protecting existing forest carbon, lengthening harvest cycles, reforestation, afforestation, and bioenergy production with product substitution. The largest potential increase in forest carbon is in the mesic Coast Range and West Cascade ecoregions. These forests are buffered by the ocean, have high soil water-holding capacity, low risk of wildfire [fire intervals average 260–400 y (18)], long carbon residence time, and potential for high carbon density. They can attain biomass up to 520 Mg C ha^{-1} (12). Although Oregon has several protected areas, they account for only 9–15% of the total forest area, so we expect it may be feasible to add carbon-protected lands with cobenefits of water protection and biodiversity.

Reforestation of recently forested areas include those areas impacted by fire and beetles. Our simulations to 2100 assume regrowth of the same species and incorporate future fire responses to climate and cyclical beetle outbreaks [70–80 y (13)]. Reforestation has the potential to increase stocks by 315 Tg C by 2100, reducing forest sector net emissions by 5% by 2100 relative to BAU management (Fig. 3). The East and West Cascades ecoregions had the highest reforestation potential, accounting for 90% of the increase (Table S5).

Afforestation of old fields within forest boundaries and nonfood/nonforage grass crops, hereafter referred to as "grass crops," had to meet minimum conditions for tree growth, and crop grid cells had to be partially forested (*SI Methods* and Table S6). These crops are not grazed or used for animal feed. Competing land uses may decrease the actual amount of area that can be afforested. We calculated the amount of irrigated grass crops (127,000 ha) that could be converted to forest, assuming success of carbon offset programs (19). By 2100, afforestation increased stocks by

Table 1. Forest carbon budget components used to compute NECB

Flux, Tg C·y^{-1}	2001–2005		2006–2010		2011–2015		2001–2015
NPP	73.64	7.59	73.57	7.58	73.57	7.58	73.60
Rh	45.67	5.11	45.38	5.07	45.19	5.05	45.41
NEP	27.97	9.15	28.19	9.12	28.39	9.11	28.18
Harvest removals	8.58	0.60	7.77	0.54	8.61	0.6	8.32
Fire emissions	2.37	0.27	1.79	0.2	0.97	0.11	1.71
NECB	17.02	9.17	18.63	9.14	18.81	9.13	18.15

Average annual values for each period, including uncertainty (95% confidence interval) in Tg C y^{-1} (multiply by 3.667 to get million tCO_2e).

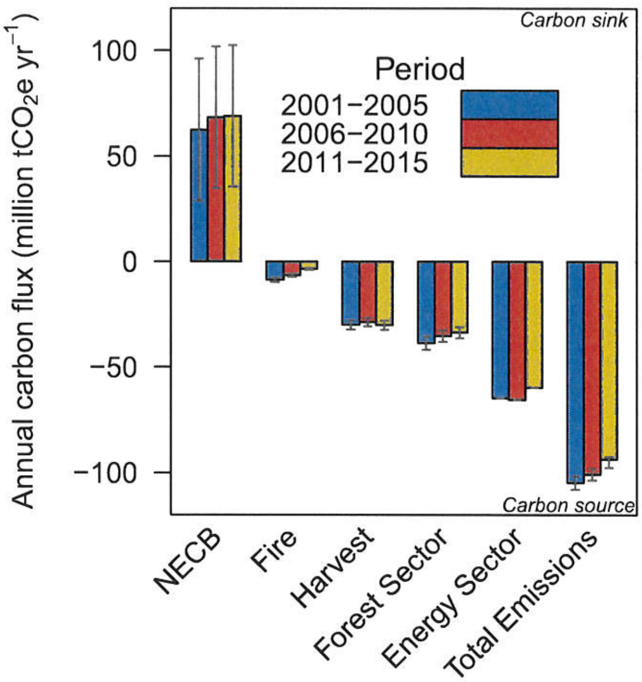


Fig. 2. Oregon's forest carbon sink and emissions from forest and energy sectors. Harvest emissions are computed by LCA. Fire and harvest emissions sum to forest sector emissions. Energy sector emissions are from the Oregon Global Warming Commission (14), minus forest-related emissions. Error bars are 95% confidence intervals (Monte Carlo analysis).

94 Tg C and cumulative NECB by 14 Tg C, and afforestation reduced forest sector GHG emissions by 1.3–1.4% in 2025, 2050, and 2100 (Fig. 3).

We quantified cobenefits of afforestation of irrigated grass crops on water availability based on data from hydrology and agricultural simulations of future grass crop area and related irrigation demand (20). Afforestation of 127,000 ha of grass cropland with Douglas fir could decrease irrigation demand by 222 and 233 billion $m^3 \cdot y^{-1}$ by 2050 and 2100, respectively. An independent estimate from measured precipitation and evapotranspiration (ET) at our mature Douglas fir and grass crop flux sites in the Willamette Valley shows the ET/precipitation fraction averaged 33% and 52%, respectively, and water balance (precipitation minus ET) averaged 910 $mm \cdot y^{-1}$ and 516 $mm \cdot y^{-1}$. Under current climate conditions, the observations suggest an increase in annual water availability of 260 billion $m^3 \cdot y^{-1}$ if 127,000 ha of the irrigated grass crops were converted to forest.

Harvest cycles in the mesic and montane forests have declined from over 120 y to 45 y despite the fact that these trees can live 500–1,000 y and net primary productivity peaks at 80–125 y (21). If harvest cycles were lengthened to 80 y on private lands and harvested area was reduced 50% on public lands, state-level stocks would increase by 17% to a total of ~3,600 Tg C and NECB would increase 2–3 Tg C y^{-1} by 2100. The lengthened harvest cycles reduced harvest by 2 Tg C y^{-1}, which contributed to higher NECB. Leakage (more harvest elsewhere) is difficult to quantify and could counter these carbon gains. However, because harvest on federal lands was reduced significantly since 1992 (NW Forest Plan), leakage has probably already occurred.

The four strategies together increased NECB by 64%, 82%, and 56% by 2025, 2050, and 2100, respectively. This reduced forest sector net emissions by 11%, 10%, and 17% over the same periods (Fig. 3). By 2050, potential increases in NECB were largest in the Coast Range (Table S5), East Cascades, and Klamath Mountains, accounting for 19%, 25%, and 42% of the total increase, whereas by 2100, they were most evident in the West Cascades, East Cascades, and Klamath Mountains.

We examined the potential for using existing harvest residue for electricity generation, where burning the harvest residue for energy emits carbon immediately (3) versus the BAU practice of leaving residues in forests to slowly decompose. Assuming half of forest residues from harvest practices could be used to replace natural gas or coal in distributed facilities across the state, they would provide an average supply of 0.75–1 Tg C y^{-1} to the year 2100 in the reduced harvest and BAU scenarios, respectively. Compared with BAU harvest practices, where residues are left to decompose, proposed bioenergy production would increase cumulative net emissions by up to 45 Tg C by 2100. Even at 50% use, residue collection and transport are not likely to be economically viable, given the distances (>200 km) to Oregon's facilities.

Discussion

Earth system models have the potential to bring terrestrial observations related to climate, vulnerability, impacts, adaptation,

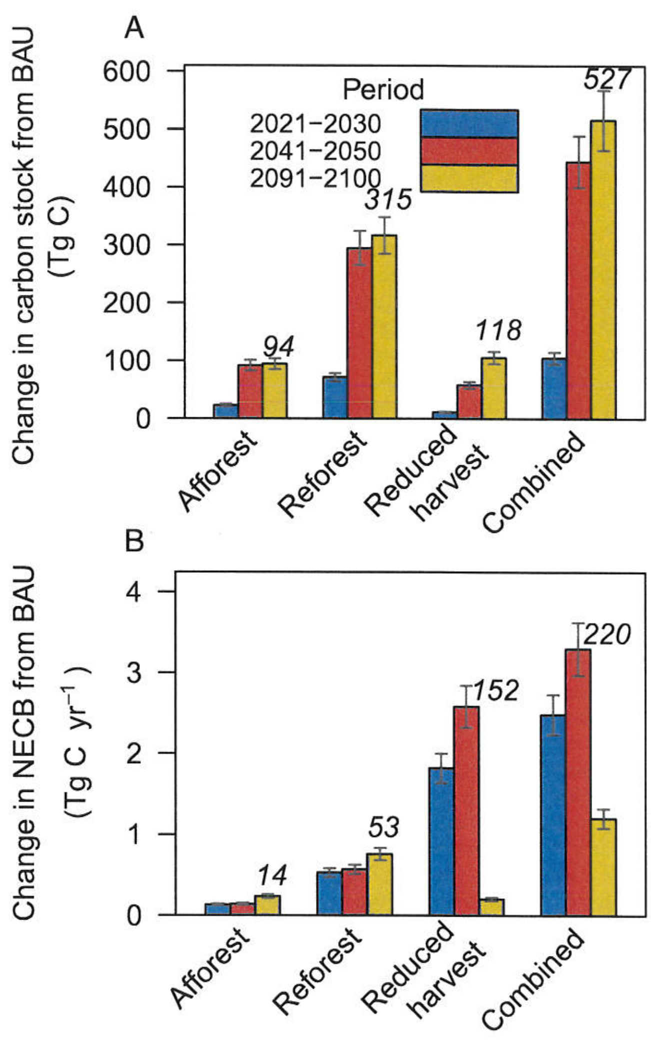


Fig. 3. Future change in carbon stocks and NECB with mitigation strategies relative to BAU management. The decadal average change in forest carbon stocks (*A*) and NECB relative to BAU (*B*) are shown. Italicized numbers over bars indicate mean forest carbon stocks in 2091–2100 (*A*) and cumulative change in NECB for 2015–2100 (*B*). Error bars are ±10%.

and mitigation into a common framework, melding biophysical with social components (22). We developed a framework to examine a suite of mitigation actions to increase forest carbon sequestration and reduce forest sector emissions under current and future environmental conditions.

Harvest-related emissions had a large impact on recent forest NECB, reducing it by an average of 34% from 2001 to 2015. By comparison, fire emissions were relatively small and reduced NECB by 12% in the Biscuit Fire year, but only reduced NECB 5–9% from 2006 to 2015. Thus, altered forest management has the potential to enhance the forest carbon balance and reduce emissions.

Future NEP increased because enhancement from atmospheric carbon dioxide outweighed the losses from fire. Lengthened harvest cycles on private lands to 80 y and restricting harvest to 50% of current rates on public lands increased NECB the most by 2100, accounting for 90% of total emissions reduction (Fig. 3 and Tables S5 and S6). Reduced harvest led to NECB increasing earlier than the other strategies (by 2050), suggesting this could be a priority for implementation.

Our afforestation estimates may be too conservative by limiting them to nonforest areas within current forest boundaries and 127,000 ha of irrigated grass cropland. There was a net loss of 367,000 ha of forest area in Oregon and Washington combined from 2001 to 2006 (23), and less than 1% of native habitat remains in the Willamette Valley due to urbanization and agriculture (24). Perhaps more of this area could be afforested.

The spatial variation in the potential for each mitigation option to improve carbon stocks and fluxes shows that the reforestation potential is highest in the Cascade Mountains, where fire and insects occur (Fig. 4). The potential to reduce harvest on public land is highest in the Cascade Mountains, and that to lengthen harvest cycles on private lands is highest in the Coast Range.

Although western Oregon is mesic with little expected change in precipitation, the afforestation cobenefits of increased water availability will be important. Urban demand for water is projected to increase, but agricultural irrigation will continue to consume much more water than urban use (25). Converting 127,000 ha of irrigated grass crops to native forests appears to be a win–win strategy, returning some of the area to forest land, providing habitat and connectivity for forest species, and easing irrigation demand. Because the afforested grass crop represents only 11% of the available grass cropland (1.18 million ha), it is not likely to result in leakage or indirect land use change. The two forest strategies combined are likely to be important contributors to water security.

Cobenefits with biodiversity were not assessed in our study. However, a recent study showed that in the mesic forests, cobenefits with biodiversity of forest species are largest on lands with harvest cycles longer than 80 y, and thus would be most pronounced on private lands (26). We selected 80 y for the harvest cycle mitigation strategy because productivity peaks at 80–125 y in this region, which coincides with the point at which cobenefits with wildlife habitat are substantial.

Habitat loss and climate change are the two greatest threats to biodiversity. Afforestation of areas that are currently grass crops would likely improve the habitat of forest species (27), as about 90% of the forests in these areas were replaced by agriculture. About 45 mammal species are at risk because of range contraction (28). Forests are more efficient at dissipating heat than grass and crop lands, and forest cover gains lead to net surface cooling in all regions south of about 45° latitude in North American and Europe (29). The cooler conditions can buffer climate-sensitive bird populations from approaching their thermal limits and provide more food and nest sites (30). Thus, the mitigation strategies of afforestation, protecting forests on public lands and lengthening harvest cycles to 80–125 y, would likely benefit forest-dependent species.

Oregon has a legislated mandate to reduce emissions, and is considering an offsets program that limits use of offsets to 8% of

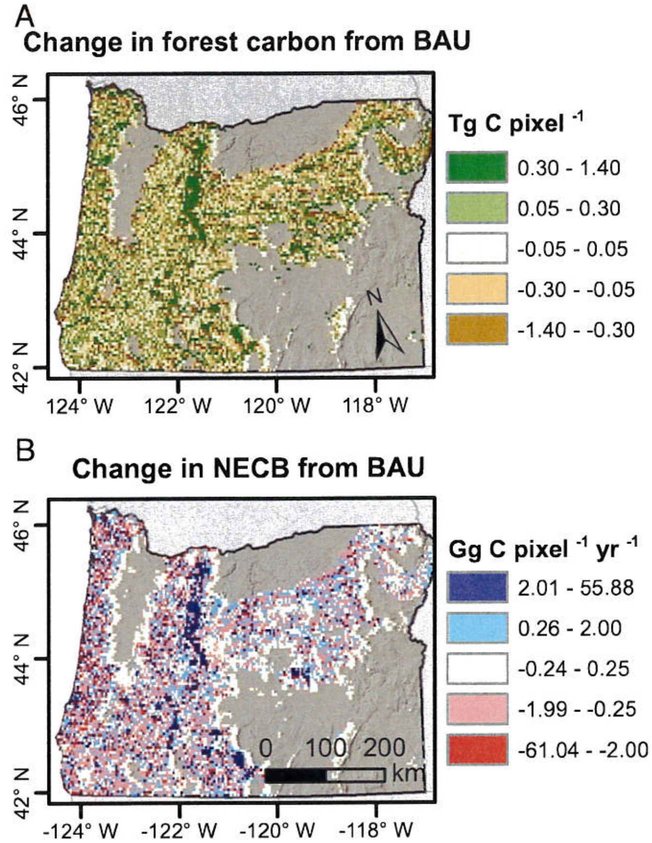


Fig. 4. Spatial patterns of forest carbon stocks and NECB by 2091–2100. The decadal average changes in forest carbon stocks (*A*) and NECB (*B*) due to afforestation, reforestation, protected areas, and lengthened harvest cycles relative to continued BAU forest management (red is increase in NECB) are shown.

the total emissions reduction to ensure that regulated entities substantially reduce their own emissions, similar to California's program (19). An offset becomes a net emissions reduction by increasing the forest carbon sink (NECB). If only 8% of the GHG reduction is allowed for forest offsets, the limits for forest offsets would be 2.1 and 8.4 million metric tCO_2e of total emissions by 2025 and 2050, respectively (Table S6). The combination of afforestation, reforestation, and reduced harvest would provide 13 million metric tCO_2e emissions reductions, and any one of the strategies or a portion of each could be applied. Thus, additionality beyond what would happen without the program is possible.

State-level reporting of GHG emissions includes the agriculture sector, but does not appear to include forest sector emissions, except for industrial fuel (i.e., utility fuel in Table S3) and, potentially, fire emissions. Harvest-related emissions should be quantified, as they are much larger than fire emissions in the western United States. Full accounting of forest sector emissions is necessary to meet climate mitigation goals.

Increased long-term storage in buildings and via product substitution has been suggested as a potential climate mitigation option. Pacific temperate forests can store carbon for many hundreds of years, which is much longer than is expected for buildings that are generally assumed to outlive their usefulness or be replaced within several decades (7). By 2035, about 75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends (31, 32). Recent analysis suggests substitution benefits of using wood versus more fossil fuel-intensive materials have been overestimated by at

AGENDA ITEM A
Attachment 08
Page 26 of 44

least an order of magnitude (33). Our LCA accounts for losses in product substitution stores (PSSs) associated with building life span, and thus are considerably lower than when no losses are assumed (4, 34). While product substitution reduces the overall forest sector emissions, it cannot offset the losses incurred by frequent harvest and losses associated with product transportation, manufacturing, use, disposal, and decay. Methods for calculating substitution benefits should be improved in other regional assessments.

Wood bioenergy production is interpreted as being carbon-neutral by assuming that trees regrow to replace those that burned. However, this does not account for reduced forest carbon stocks that took decades to centuries to sequester, degraded productive capacity, emissions from transportation and the production process, and biogenic/direct emissions at the facility (35). Increased harvest through proposed thinning practices in the region has been shown to elevate emissions for decades to centuries regardless of product end use (36). It is therefore unlikely that increased wood bioenergy production in this region would decrease overall forest sector emissions.

Conclusions

GHG reduction must happen quickly to avoid surpassing a 2 °C increase in temperature since preindustrial times. Alterations in forest management can contribute to increasing the land sink and decreasing emissions by keeping carbon in high biomass forests, extending harvest cycles, reforestation, and afforestation. Forests are carbon-ready and do not require new technologies or infrastructure for immediate mitigation of climate change. Growing forests for bioenergy production competes with forest carbon sequestration and does not reduce emissions in the next decades (10). BECCS requires new technology, and few locations have sufficient geological storage for CO_2 at power facilities with high-productivity forests nearby. Accurate accounting of forest carbon in trees and soils, NECB, and historic harvest rates, combined with transparent quantification of emissions from the wood product process, can ensure realistic reductions in forest sector emissions.

As states and regions take a larger role in implementing climate mitigation steps, robust forest sector assessments are urgently needed. Our integrated approach of combining observations, an LCA, and high-resolution process modeling (4-km grid vs. typical 200-km grid) of a suite of potential mitigation actions and their effects on forest carbon sequestration and emissions under changing climate and CO_2 provides an analysis framework that can be applied in other temperate regions.

Materials and Methods

Current Stocks and Fluxes. We quantified recent forest carbon stocks and fluxes using a combination of observations from FIA; Landsat products on forest type, land cover, and fire risk; 200 intensive plots in Oregon (37); and a wood decomposition database. Tree biomass was calculated from species-specific allometric equations and ecoregion-specific wood density. We estimated ecosystem carbon stocks, NEP (photosynthesis minus respiration), and NECB (NEP minus losses due to fire or harvest) using a mass-balance approach (36, 38) (Table 1 and *SI Materials and Methods*). Fire emissions were computed from the Monitoring Trends in Burn Severity database, biomass data, and region-specific combustion factors (15, 39) (*SI Materials and Methods*).

Future Projections and Model Description. Carbon stocks and NEP were quantified to the years 2025, 2050, and 2100 using CLM4.5 with physiological parameters for 10 major forest species, initial forest biomass (36), and future climate and atmospheric carbon dioxide as input (Institut Pierre Simon Laplace climate system model downscaled to 4 km × 4 km, representative concentration pathway 8.5). CLM4.5 uses 3-h climate data, ecophysiological characteristics, site physical characteristics, and site history to estimate the daily fluxes of carbon, nitrogen, and water between the atmosphere, plant state variables, and litter and soil state variables. Model components are biogeophysics, hydrological cycle, and biogeochemistry. This model version does not include a dynamic vegetation model to simulate resilience and establishment following disturbance. However, the effect of regeneration lags on forest carbon is not particularly strong for the long disturbance intervals in this study (40). Our plant functional type (PFT) parameterization for 10 major forest species rather than one significantly improves carbon modeling in the region (41).

Forest Management and Land Use Change Scenarios. Harvest cycles, reforestation, and afforestation were simulated to the year 2100. Carbon stocks and NEP were predicted for the current harvest cycle of 45 y compared with simulations extending it to 80 y. Reforestation potential was simulated over areas that recently suffered mortality from harvest, fire, and 12 species of beetles (13). We assumed the same vegetation regrew to the maximum potential, which is expected with the combination of natural regeneration and planting that commonly occurs after these events. Future BAU harvest files were constructed using current harvest rates, where county-specific average harvest and the actual amounts per ownership were used to guide grid cell selection. This resulted in the majority of harvest occurring on private land (70%) and in the mesic ecoregions. Beetle outbreaks were implemented using a modified mortality rate of the lodgepole pine PFT with 0.1% y^{-1} biomass mortality by 2100.

For afforestation potential, we identified areas that are within forest boundaries that are not currently forest and areas that are currently grass crops. We assumed no competition with conversion of irrigated grass crops to urban growth, given Oregon's land use laws for developing within urban growth boundaries. A separate study suggested that, on average, about 17% of all irrigated agricultural crops in the Willamette Valley could be converted to urban area under future climate; however, because 20% of total cropland is grass seed, it suggests little competition with urban growth (25).

Landsat observations (12,500 scenes) were processed to map changes in land cover from 1984 to 2012. Land cover types were separated with an unsupervised K-means clustering approach. Land cover classes were assigned to an existing forest type map (42). The CropScape Cropland Data Layer (CDL 2015, https://nassgeodata.gmu.edu/CropScape/) was used to distinguish nonforage grass crops from other grasses. For afforestation, we selected grass cropland with a minimum soil water-holding capacity of 150 mm and minimum precipitation of 500 mm that can support trees (43).

Afforestation Cobenefits. Modeled irrigation demand of grass seed crops under future climate conditions was previously conducted with hydrology and agricultural models, where ET is a function of climate, crop type, crop growth state, and soil-holding capacity (20) (Table S7). The simulations produced total land area, ET, and irrigation demand for each cover type. Current grass seed crop irrigation in the Willamette Valley is 413 billion $m^3 \cdot y^{-1}$ for 238,679 ha and is projected to be 412 and 405 billion m^3 in 2050 and 2100 (20) (Table S7). We used annual output from the simulations to estimate irrigation demand per unit area of grass seed crops (1.73, 1.75, and 1.84 million $m^3 \cdot ha^{-1}$ in 2015, 2050, and 2100, respectively), and applied it to the mapped irrigated crop area that met conditions necessary to support forests (Table S7).

LCA. Decomposition of wood through the product cycle was computed using an LCA (8, 10). Carbon emissions to the atmosphere from harvest were calculated annually over the time frame of the analysis (2001–2015). The net carbon emissions equal NECB plus total harvest minus wood lost during manufacturing and wood decomposed over time from product use. Wood industry fossil fuel emissions were computed for harvest, transportation, and manufacturing processes. Carbon credit was calculated for wood product storage, substitution, and internal mill recycling of wood losses for bioenergy.

Products were divided into sawtimber, pulpwood, and wood and paper products using published coefficients (44). Long-term and short-term products were assumed to decay at 2% and 10% per year, respectively (45). For product substitution, we focused on manufacturing for long-term structures (building life span >30 y). Because it is not clear when product substitution started in the Pacific Northwest, we evaluated it starting in 1970 since use of concrete and steel for housing was uncommon before 1965. The displacement value for product substitution was assumed to be 2.1 Mg fossil C/Mg C wood use in long-term structures (46), and although it likely fluctuates over time, we assumed it was constant. We accounted for losses in product substitution associated with building replacement (33) using a loss rate of 2% per year (33), but ignored leakage related to fossil C use by other sectors, which may result in more substitution benefit than will actually occur.

The general assumption for modern buildings, including cross-laminate timber, is they will outlive their usefulness and be replaced in about 30 y (7). By 2035, ~75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends, resulting in threefold as many buildings as there are now [2005 baseline (31, 32)]. The loss of

ENVIRONMENTAL SCIENCES

SUSTAINABILITY SCIENCE

the PSS is therefore PSS multiplied by the proportion of buildings lost per year (2% per year).

To compare the NECB equivalence to emissions, we calculated forest sector and energy sector emissions separately. Energy sector emissions ["in-boundary" state-quantified emissions by the Oregon Global Warming Commission (14)] include those from transportation, residential and commercial buildings, industry, and agriculture. The forest sector emissions are cradle-to-grave annual carbon emissions from harvest and product emissions, transportation, and utility fuels (Table S3). Forest sector utility fuels were subtracted from energy sector emissions to avoid double counting.

Uncertainty Estimates. For the observation-based analysis, Monte Carlo simulations were used to conduct an uncertainty analysis with the mean and SDs for NPP and Rh calculated using several approaches (36) (*SI Materials and Methods*). Uncertainty in NECB was calculated as the combined uncertainty of NEP, fire emissions (10%), harvest emissions (7%), and land cover estimates (10%) using the propagation of error approach. Uncertainty in CLM4.5 model simulations and LCA were quantified by combining the uncertainty in the observations used to evaluate the model, the uncertainty in input datasets (e.g., remote sensing), and the uncertainty in the LCA coefficients (41).

Model input data for physiological parameters and model evaluation data on stocks and fluxes are available online (37).

ACKNOWLEDGMENTS. We thank Dr. Thomas Hilker (deceased) for producing the grass data layer, and Carley Lowe for Fig. 1. This research was supported by the US Department of Energy (Grant DE-SC0012194) and Agriculture and Food Research Initiative of the US Department of Agriculture National Institute of Food and Agriculture (Grants 2013-67003-20652, 2014-67003-22065, and 2014-35100-22066) for our North American Carbon Program studies, "Carbon cycle dynamics within Oregon's urban-suburban-forested-agricultural landscapes," and "Forest die-off, climate change, and human intervention in western North America."

1. Canadell JG, Raupach MR (2008) Managing forests for climate change mitigation. *Science* 320:1456–1457.
2. Holtsmark B (2013) The outcome is in the assumptions: Analyzing the effects on atmospheric CO2 levels of increased use of bioenergy from forest biomass. *Glob Change Biol Bioenergy* 5:467–473.
3. Repo A, Tuomi M, Liski J (2011) Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues. *Glob Change Biol Bioenergy* 3:107–115.
4. Schlamadinger B, Marland G (1996) The role of forest and bioenergy strategies in the global carbon cycle. *Biomass Bioenergy* 10:275–300.
5. Heck V, Gerten D, Lucht W, Popp A (2018) Biomass-based negative emissions difficult to reconcile with planetary boundaries. *Nat Clim Chang* 8:151–155.
6. Field CB, Mach KJ (2017) Rightsizing carbon dioxide removal. *Science* 356:706–707.
7. Tollefson J (2017) The wooden skyscrapers that could help to cool the planet. *Nature* 545:280–282.
8. Law BE, Harmon ME (2011) Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. *Carbon Manag* 2: 73–84.
9. Law BE (2014) Regional analysis of drought and heat impacts on forests: Current and future science directions. *Glob Change Biol* 20:3595–3599.
10. Hudiburg TW, Luyssaert S, Thornton PE, Law BE (2013) Interactive effects of environmental change and management strategies on regional forest carbon emissions. *Environ Sci Technol* 47:13132–13140.
11. Keith H, Mackey BG, Lindenmayer DB (2009) Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proc Natl Acad Sci USA* 106:11635–11640.
12. Law B, Waring R (2015) Carbon implications of current and future effects of drought, fire and management on Pacific Northwest forests. *For Ecol Manage* 355:4–14.
13. Berner LT, Law BE, Meddens AJ, Hicke JA (2017) Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States (2003–2012). *Environ Res Lett* 12:065005.
14. Oregon Global Warming Commission (2017) Biennial Report to the Legislature (Oregon Global Warming Commission, Salem, OR).
15. Campbell J, Donato D, Azuma D, Law B (2007) Pyrogenic carbon emission from a large wildfire in Oregon, United States. *J Geophys Res Biogeosci* 112:G04014.
16. King AW, Hayes DJ, Huntzinger DN, West TO, Post WM (2012) North American carbon dioxide sources and sinks: Magnitude, attribution, and uncertainty. *Front Ecol Environ* 10:512–519.
17. Wilson BT, Woodall CW, Griffith DM (2013) Imputing forest carbon stock estimates from inventory plots to a nationally continuous coverage. *Carbon Balance Manag* 8:1.
18. Reilly MJ, et al. (2017) Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). *Ecosphere* 8:e01695.
19. Anderson CM, Field CB, Mach KJ (2017) Forest offsets partner climate-change mitigation with conservation. *Front Ecol Environ* 15:359–365.
20. Willamette Water 2100 Explorer (2017) Assessing water futures under alternative climate and management scenarios: Agricultural water demand, crop and irrigation decisions. Available at explorer.bee.oregonstate.edu/Topic/WW2100/AgSummaries.aspx. Accessed November 13, 2017.
21. Hudiburg T, et al. (2009) Carbon dynamics of Oregon and Northern California forests and potential land-based carbon storage. *Ecol Appl* 19:163–180.
22. Bonan GB, Doney SC (2018) Climate, ecosystems, and planetary futures: The challenge to predict life in Earth system models. *Science* 359:eaam8328.
23. Riitters KH, Wickham JD (2012) Decline of forest interior conditions in the conterminous United States. *Sci Rep* 2:653.
24. Noss RF, LaRoe ET, Scott JM (1995) *Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation* (US Department of the Interior, National Biological Service, Washington, DC).
25. Jaeger W, Plantinga A, Langpap C, Bigelow D, Moore K (2017) *Water, Economics and Climate Change in the Willamette Basin* (Oregon State University Extension Service, Corvallis, OR).
26. Kline JD, et al. (2016) Evaluating carbon storage, timber harvest, and habitat possibilities for a Western Cascades (USA) forest landscape. *Ecol Appl* 26:2044–2059.
27. Matthews S, O'Connor R, Plantinga AJ (2002) Quantifying the impacts on biodiversity of policies for carbon sequestration in forests. *Ecol Econ* 40:71–87.
28. Arnold S, Kagan J, Taylor B (2000) Summary of current status of Oregon's biodiversity. *Oregon State of the Environment Report* (Oregon Progress Board, Salem, OR), pp 121–126.
29. Bright RM, et al. (2017) Local temperature response to land cover and management change driven by non-radiative processes. *Nat Clim Chang* 7:296–302.
30. Betts M, Phalan B, Frey S, Rousseau J, Yang Z (2017) Old-growth forests buffer climate-sensitive bird populations from warming. *Diversity Distrib*, 10.1111/ddi.12688.
31. Architecture 2030(2017) The 2030 Challenge. Available at architecture2030.org. Accessed October 23, 2017.
32. Oliver CD, Nassar NT, Lippke BR, McCarter JB (2014) Carbon, fossil fuel, and biodiversity mitigation with wood and forests. *J Sustain For* 33:248–275.
33. Harmon ME, Moreno A, Domingo JB (2009) Effects of partial harvest on the carbon stores in Douglas-fir/western hemlock forests: A simulation study. *Ecosystems (N Y)* 12:777–791.
34. Lippke B, et al. (2011) Life cycle impacts of forest management and wood utilization on carbon mitigation: Knowns and unknowns. *Carbon Manag* 2:303–333.
35. Gunn JS, Ganz DJ, Keeton WS (2011) Biogenic vs. geologic carbon emissions and forest biomass energy production. *GCB Bioenergy* 4:239–242.
36. Hudiburg TW, Law BE, Wirth C, Luyssaert S (2011) Regional carbon dioxide implications of forest bioenergy production. *Nat Clim Chang* 1:419–423.
37. Law BE, Berner LT (2015) *NACP TERRA-PNW: Forest Plant Traits, NPP, Biomass, and Soil Properties, 1999-2014* (ORNL DAAC, Oak Ridge, TN).
38. Law BE, Hudiburg TW, Luyssaert S (2013) Thinning effects on forest productivity: Consequences of preserving old forests and mitigating impacts of fire and drought. *Plant Ecol Divers* 6:73–85.
39. Meigs G, Donato D, Campbell J, Martin J, Law B (2009) Forest fire impacts on carbon uptake, storage, and emission: The role of burn severity in the Eastern Cascades, Oregon. *Ecosystems (N Y)* 12:1246–1267.
40. Harmon ME, Marks B (2002) Effects of silvicultural practices on carbon stores in Douglas-fir western hemlock forests in the Pacific Northwest, USA: Results from a simulation model. *Can J For Res* 32:863–877.
41. Hudiburg TW, Law BE, Thornton PE (2013) Evaluation and improvement of the community land model (CLM4) in Oregon forests. *Biogeosciences* 10:453–470.
42. Ruefenacht B, et al. (2008) Conterminous U.S. and Alaska forest type mapping using forest inventory and analysis data. *Photogramm Eng Remote Sens* 74:1379–1388.
43. Peterman W, Bachelet D, Ferschweiler K, Sheehan T (2014) Soil depth affects simulated carbon and water in the MC2 dynamic global vegetation model. *Ecol Modell* 294:84–93.
44. Smith JE, Heath L, Skog KE, Birdsey R (2006) Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States (US Department of Agriculture, Forest Service, Northeastern Research Station, Newtown Square, PA), General Technical Report NE-343.
45. Harmon ME, Harmon JM, Ferrell WK, Brooks D (1996) Modeling carbon stores in Oregon and Washington forest products: 1900–1992. *Clim Change* 33:521–550.
46. Sathre R, O'Connor J (2010) Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environ Sci Policy* 13:104–114.

Received: 18 February 2016 | Revised: 17 July 2016 | Accepted: 26 August 2016
DOI 10.1002/eco.1790

WILEY

SPECIAL ISSUE PAPER

Summer streamflow deficits from regenerating Douglas-fir forest in the Pacific Northwest, USA

Timothy D. Perry | Julia A. Jones

Geography, College of Earth, Ocean, and Atmospheric Sciences, Oregon State University, Corvallis, OR 97331, USA

Correspondence
Julia A. Jones, Geography, College of Earth, Ocean, and Atmospheric Sciences, Oregon State University, Corvallis, OR 97331, USA.
Email: jonesj@geo.oregonstate.edu

Abstract

Despite controversy about effects of plantation forestry on streamflow, streamflow response to forest plantations over multiple decades is not well understood. Analysis of 60-year records of daily streamflow from eight paired-basin experiments in the Pacific Northwest of the United States (Oregon) revealed that the conversion of old-growth forest to Douglas-fir plantations had a major effect on summer streamflow. Average daily streamflow in summer (July through September) in basins with 34- to 43-year-old plantations of Douglas-fir was 50% lower than streamflow from reference basins with 150- to 500-year-old forests dominated by Douglas-fir, western hemlock, and other conifers. Study plantations are comparable in terms of age class, treatments, and growth rates to managed forests in the region. Young Douglas-fir trees, which have higher sapwood area, higher sapflow per unit of sapwood area, higher concentration of leaf area in the upper canopy, and less ability to limit transpiration, appear to have higher rates of evapotranspiration than old trees of conifer species, especially during dry summers. Reduced summer streamflow in headwater basins with forest plantations may limit aquatic habitat and exacerbate stream warming, and it may also alter water yield and timing in much larger basins. Legacies of past forest management or extensive natural disturbances may be confounded with effects of climate change on streamflow in large river basins. Continued research is needed using long-term paired-basin studies and process studies to determine the effects of forest management on streamflow deficits in a variety of forest types and forest management systems.

KEYWORDS

climate change, native forests, plantations, stationarity, succession, water scarcity

1 | INTRODUCTION

Widespread evidence that streamflow is declining in major rivers in the United States and globally has raised concerns about water scarcity (Adam, Hamlet, & Lettenmaier, 2009; Dai, Qian, Trenberth, & Milliman, 2009; Luce & Holden, 2009; Vörösmarty, Green, Salisbury, & Lammers, 2000). Climate change and variability are implicated as causes of many streamflow trends (Lins & Slack, 1999, 2005; McCabe & Wolock, 2002; Mote et al., 2003; Hodgkins, Dudley, & Huntington, 2003, 2005; Stewart, Cayan, & Dettinger, 2004, 2005; Nolin & Daly, 2006; Hamlet & Lettenmaier, 2007; Barnett et al., 2008; Jefferson, Nolin, Lewis, & Tague, 2008; Lara, Villalba, & Urrutia, 2008; Dai et al., 2009; Kennedy, Garen, & Koch, 2009; Jones, 2011). However, large-scale plantation forestry, often using non-native tree species, is expanding in much of the temperate zone on Earth, despite widespread evidence that intensive forestry reduces water yield (Cornish & Vertessy, 2001; Andréassian, 2004; Brown, Zhang, McMahon, Western, & Vertessy, 2005, Farley, Jobbágy, & Jackson, 2005; Sun et al., 2006; Little, Lara, McPhee, & Urrutia, 2009). Water yield reductions are greater in older plantations, during dry seasons, and in arid regions (Andréassian, 2004; Brown et al., 2005; Farley et al., 2005; Sun et al., 2006). Yet, downstream effects of forestry are debated (van Dijk & Keenan, 2007).

Despite general studies of water partitioning in forested basins (e.g., Budyko, 1974, Zhang, Dawes, & Walker, 2001, Jones et al., 2012), it is unclear how streamflow varies during forest succession, relative to tree species, age, or growth rates in native forest and forest plantations (Creed et al., 2014). In the Pacific Northwest of the United States, forest plantations have reduced summer streamflow relative to mature and old-growth forest (Hicks, Beschta, & Harr, 1991; Jones & Post, 2004). However, the magnitude, duration, causes, and consequences of summer water deficits associated with forest plantations are not well understood.

In the Pacific Northwest, large areas of old-growth forest have been converted to forest plantations. We examined how changes in forest structure and composition have affected streamflow using multiple paired-basin experiments in western and southwestern Oregon, where regenerating forests are currently aged 40 to 50 years, and reference forests are aged 150 to 500 years. Many studies have reported on these experiments, including vegetation ecology (e.g., Marshall & Waring, 1984; Halpern, 1989; Halpern & Franklin, 1990; Halpern & Spies, 1995; Lutz & Halpern, 2006; Halpern & Lutz, 2013) and hydrology (e.g., Rothacher, 1970; Harr, Fredriksen, & Rothacher, 1979; Harr & McCorison, 1979; Harr, Levno, & Mersereau, 1982; Hicks et al., 1991; Jones & Grant, 1996, Jones, 2000; Jones & Post, 2004, Perkins & Jones, 2008; Jones & Perkins, 2010; Jennings & Jones, 2015). We asked:

1. How has daily streamflow changed over the past half-century in reference basins with 150- to 500-year-old forest?

TABLE 1 Name and abbreviation, area, elevation range, natural vegetation and vegetation age when streamflow records began, streamflow gaging method and record length, harvest treatment, logging methods, and treatment dates for basins used in this study

Basin name	Area (ha)	Elevation range (m)	Natural vegetation	Streamflow record length, instrumentation[b]	Treatment, date[a]	Logging method
Coyote 1 COY 1	69.2	750–1,065	Mixed conifer	1963–81 V; 2001–present V	Roads 1970; 50% overstory selective cut, 1971	Tractor yarded
Coyote 2 COY 2	68.4	760–1,020	Mixed conifer	1963–81 V; 2001–present V	Permanent roads 1970; 30% 2- to 3-ha patch cuts, 1971	16% high-lead cable yarded; 14% tractor yarded.
Coyote 3 COY 3	49.8	730–960	Mixed conifer	1963–81 V; 2001–present V	Permanent roads 1970; 100%; clearcut 1971	77% high-lead cable yarded; 23% tractor yarded.
Coyote 4 COY 4	48.6	730–930	Mixed conifer	1963–81 V; 2001–present V	Reference	N/A
Andrews 1 AND 1	95.9	460–990	450- to 500-year-old Douglas-fir forest	1952–present (1952–present T [rebuilt 1956]; 1999–present SV)	100% clearcut 1962–1966, broadcast burn 1966	100% skyline yarded
Andrews 2 AND 2	60.7	530–1,070	450- to 500-year-old Douglas-fir forest	1952–present (1952–present T; 1999–present SV)	Reference	N/A
Andrews 3 AND 3	101.2	490–1,070	450- to 500-year-old Douglas-fir forest	1952–2005 T; 1999–present SV	Roads 1959; 25% patch cut 1962, broadcast burn 1963	25% high-lead cable yarded
Andrews 6 AND 6	13.0	863–1,013	130- to 450-year-old Douglas-fir forest	1964–present; (1964–1997 H; 1997–present T; 1998 present SV)	Roads, 1974; 100% clearcut 1974; broadcast burn 1975	90% high-lead cable yarded; 10% tractor yarded
Andrews 7 AND 7	15.4	908–1,097	130- to 450-year-old Douglas-fir forest	1964–1987; 1995–present (1964–1997 H; 1997–present T; 1998–present SV)	Roads 1974; 60% shelterwood cut 1974; remaining overstory cut 1984; broadcast burn lower half of basin 1975; 12% basal area thin 2001	40% skyline yarded; 60% tractor yarded.
Andrews 8 AND 8	21.4	955–1,190	130- to 450-year-old Douglas-fir forest	1964–present (1964–1987 H; 1987 present T; 1973–1979 SV, 1997–present SV)	Reference	N/A
Andrews 9 AND 9	9	425–700	130- to 450-year-old Douglas-fir forest	1969–present (1969–1973 H; 1973 present T; 1973–1979 SV, 1997 present SV)	Reference	N/A
Andrews 10 AND 10	10	425–700	130- to 450-year-old Douglas-fir forest	1969–present (1969–1973 H; 1973 present T; 1973–1979 SV, 1997–present SV)	100% clear-cut 1975; no burn	100% high-lead cable yarded

Sources: Harr et al., 1979; Rothacher, 1965; Harr et al., 1982; Rothacher, Dyrness, & Fredriksen, 1967; Jones & Post, 2004.

[a]Broadcast burns were controlled burns over the cut area intended to consume logging debris.

[b]H = H-flume; T = trapezoidal flume; V = V-notch weir or plate. Summer V-notch weirs (SV) have been used for improved discharge measurements over the following periods: since 1999 at Andrews 1, 2, and 3; since 1998 at Andrews 6, 7, and 8; and from 1969 to 1973 and since 1997 at Andrews 9 and 10.

2. What are the trends in daily streamflow over 40- to 50-year periods, from basins with regenerating forests compared to reference basins?
3. How are changes in summer streamflow related to forest structure and composition in mature and old-growth forests versus forest plantations?

2 | STUDY SITE

The study examined streamflow changes in eight pairs of treated/reference basins in five paired-basin studies. Five of the basin pairs (eight basins) were located in the H.J. Andrews Experimental Forest (122°15′W, 44°12′N) in the Willamette National Forest. Three basin pairs (four basins) were located at Coyote Creek in the South Umpqua Experimental Forest (122°42′W, 43°13′N) in the Umpqua National Forest (Table 1; Figure 1). Basins are identified as Andrews 1, 2, etc. = AND 1, 2, etc.; Coyote 1, 2, etc. = COY 1, 2, etc. (Table 1).

The geology of the study basins is composed of highly weathered Oligocene tuffs and breccias that are prone to mass movements. The upper elevation portion of the Andrews Forest (above ~800 m, AND 6, AND 7, AND 8) is underlain by Miocene andesitic basalt lava flows (Dyrness, 1967; Swanson & James, 1975; Swanson & Swanston, 1977). Soils are loamy, well-drained, and moderately to highly permeable, with considerable variation in depth and rock content (Rothacher, 1969; Dyrness, 1969; Dyrness & Hawk, 1972).

The Andrews Forest ranges from 430 to 1,600 m elevation; study basins range from 430 to 1,100 m elevation (Table 1). Area-averaged slope gradients are >60% at low elevation (AND 1, AND 2, AND 3, AND 9, AND 10) and 30% at high elevation (AND 6, AND 7, AND 8). Mean daily temperature ranges from 2°C (December) to 20°C (July) at 430 m and from 1°C (December) to 17°C (July) at 1300 m. Mean annual precipitation is 2300 mm, >75% of precipitation falls between November and April, and actual evapotranspiration averages 45% of precipitation. The South Umpqua Experimental Forest (Coyote Creek basins) ranges from 730 to 1065 m elevation. Most slope gradients are <40% (Arthur, 2007). Mean daily temperature (at USHCN station OR356907, 756 m elevation, 30 km SE of Coyote Creek) ranges from 3°C (December) to 20°C (July). Mean annual precipitation (at OR356907) is 1,027 mm, >80% of precipitation falls between November and April, and actual evapotranspiration averages 45% of precipitation.

Study basins are located along a gradient of seasonal snow depth and duration (Harr, 1981, 1986). At high elevation (>800 m, AND 6, AND 7, and AND 8), average snowpack water equivalent on April 30 exceeds 700 mm (30% of annual precipitation), and snow may persist for 6 months, whereas at low elevation (<700 m, AND 9, AND 10), snow rarely persists more than 1–2 weeks and usually melts within 1–2 days; peak snowpack water equivalent is ~2% of precipitation

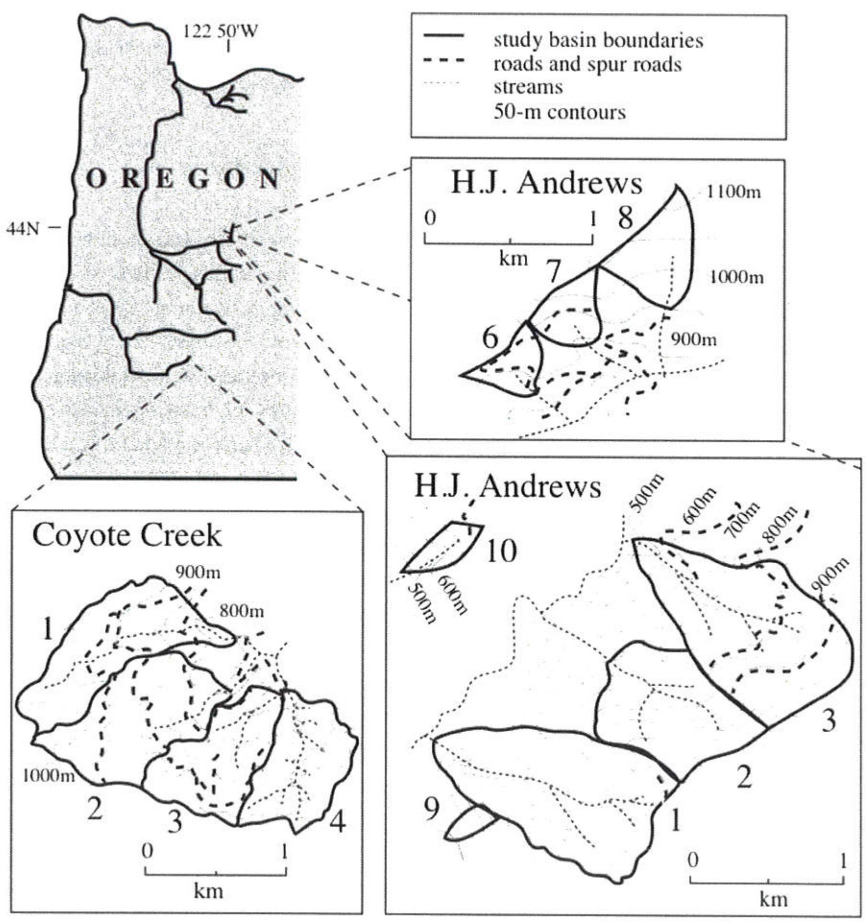


FIGURE 1 Location of study basins in western Oregon

(Harr et al., 1979; Harr & McCorison, 1979; Harr et al., 1982; Perkins & Jones, 2008). Snow at the South Umpqua Experimental Forest (Coyote Creek) usually melts within 1–2 weeks.

Vegetation at the Andrews Forest is Douglas-fir/western hemlock forest. Mature and old-growth forest regenerated after wildfires in the early 1500s and mid-1800s (Weisberg & Swanson, 2003, Tepley, 2010, Tepley, Swanson, & Spies, 2013). Overstory canopy cover is 70% to 80% and leaf area index is >8 (Dyrness & Hawk, 1972; Marshall & Waring, 1986; Lutz & Halpern, 2006). Vegetation at the South Umpqua Experimental Forest is mixed conifer (Douglas-fir, white fir, incense cedar, sugar pine), and overstory canopy cover is 70% to 80% (Anderson et al., 2013).

At the Andrews Forest, the first paired-basin experiment began in 1952 (AND 1, 2, 3); a second paired basin experiment began in 1963 (AND 6, 7, 8), and a third paired-basin experiment began in 1968 (AND 9, 10), with continuous records except at AND 7 (Table 1). Pre-treatment periods exceeded 7 years in all cases and were 10 years for AND 1/2, AND 6/8, and AND 7/8. Streamflow instrumentation changed in some basins over the period of record (Table 1). Because of the timing of instrumentation changes at AND 9/10, AND 2 is used as the reference basin for AND 10 (see Supporting Information). At the South Umpqua Experimental Forest, the Coyote Creek paired-basin experiment began in 1963 (Table 1). The pre-treatment period was 7 years. Despite a break in the record from 1981 to 2000, streamflow instrumentation at Coyote Creek has not changed (M. Jones, personal communication).

3 | METHODS

This study examined changes in daily average streamflow and its relationship to climate and forest structure and species composition in paired basins. Climate, vegetation, and streamflow have been measured for multiple decades at the Andrews Forest and Coyote Creek (see Supporting Information). Tree-level vegetation data were used to calculate basal area for all species, proportions of basal area for major species, and size class distributions.

Daily streamflow data for the period of record were used to calculate the change in streamflow by day of water year utilizing the method developed by Jones and Post (2004). R, the logarithm of the ratio of daily streamflow at the treated basin T and reference (control) basin C for year y and day d was calculated following Eberhardt and Thomas (1991) as

$$R_{y,d} = \ln\left(\frac{T_{y,d}}{C_{y,d}}\right). \tag{1}$$

The value M_{pd} was defined as the mean of R on day d for all years y in each period p.

The percent difference $\Delta_{p,d}$ between the treated: reference ratio of streamflow on day d in the post-treatment period p compared to $M_{p,d}$ in the pre-treatment period ($M_{p=0,d}$), was:

$$\Delta_{p,d} = 100\left[e^{(M_{p,d}-M_{0,d})}-1\right] \tag{2}$$

The 15-day smoothed percent change in daily streamflow, S, was calculated for all days d in each period p.

The smoothed daily percent difference S_{pd} was averaged for 5-year post-treatment periods and plotted as a function of day of the water year. S_{pd} also was summed by month and plotted as a function of time (year). Percent changes in daily streamflow were calculated for eight treated/reference basin pairs: COY 1/4, COY 2/4, COY 3/4, AND 1/2, AND 3/2, AND 6/8, AND 7/8, and AND 10/2. The significance of percent changes was assessed based on comparison with the 15-day smoothed values of the pre-treatment standard error of P_{pd}.

A daily soil water balance was created for AND 2 based on mean daily values of precipitation and discharge, daily evapotranspiration estimated from S_{pd} (Jones & Post, 2004), and mean daily snow water equivalent modeled in Perkins and Jones (2008). In addition, long-term trends in streamflow were calculated for each day of the water year from the beginning of the record to 1996, for AND 2, 8, and 9, following Hatcher and Jones (2013; see Supporting Information).

Flow percentiles were calculated for each gage record, and the numbers of days of flow below each percentile were tallied by water year. The difference in numbers of days below selected percentiles between the treated and reference basin for 1995 to 2005 was calculated and compared to summer discharge at the reference basin for 100% treated/reference pairs.

4 | RESULTS

The structure and composition of native mature and old-growth forest in reference basins varied, reflecting wildfire history, but was stable over the study period. Basal area ranged from 66 to 89 m^2/ha depending on the basin and the year (Table 2). Douglas-fir (*Pseudotsuga menziesii*) was the dominant species, representing 55 to more than 90% of basal area, with varying amounts of western hemlock (*Tsuga heterophylla*) and western redcedar (*Thuja plicata*) in AND 2 and AND 8, and California incense cedar (*Calocedrus decurrens*) and white fir (*Abies concolor*) in COY 4 (Table 2). Trees in AND 2 (N-facing) and AND 8 (upper elevation) were large, with weighted mean stem diameter of roughly 0.66 m. In contrast, trees were smaller on the low elevation, SW-facing, relatively hot, dry slopes of AND 9, and the mid-elevation COY 4 in southwest Oregon, with mean diameter of just over 0.3 m (Table 2). Stem density ranged from 87 stems per hectare at the N-facing AND 2 to over 400 stems per hectare at the SW-facing AND 9. Over a 25-year period, stem density and basal area were stable in AND 2, although there was a slight net loss of Douglas-fir and a gain of western hemlock (Table 2). The size-class distributions of Douglas-fir reveal moderate-severity historical fire in AND 2 and moderate to high-severity fire AND 8 in the mid-1800s, which produced cohorts of regenerating Douglas-fir (Figure 2).

TABLE 2 Vegetation characteristics of the study basins, sampled over the period 1981 to 2011

					Basal area								Stem density (stems per hectare)	
					(m^2/ha)	As %								
Watershed	*N* of plots	Plot size (m^2)	Year	Age	All	PSME	TSHE	THPL	ABCO	CADE	PILA	Other[a]	All	PSME
Treated patches														
AND 1	132	250	2007	40	33 ± 14	85	3	1	0	0	0	11	1,454	919
AND 3	61	250	2007	43	35 ± 12	80	11	2	0	0	0	7	1,857	621
AND 6	22	250	2008	34	35 ± 9	77	11	9	0	0	0	3	1,107	699
AND 7	24	250	2008	24	23 ± 10	70	9	4	0	0	0	17	900	551
AND 10	36	150	2010	35	27 ± 12	81	4	2	0	0	0	13	893	437
COY 1[be]	--[f]	--[f]	2011	35–200[g]	66	56	5	0	17	12	5	5	992	194
COY 2[c]	4	150	2006	35	31 ± 12	82	0	0	0	13	0	5	1,733	1,150
COY 3[c]	4	150	2006	35	45 ± 13	80	0	0	0	10	0	10	1,533	1,083
Reference														
AND 2	67	250	1981	150–475[d]	69 ± 29	70	24	2	0	0	0	4	262	67
	67	250	2006	175–500[d]	72 ± 29	65	29	2	0	0	0	4	438	87
AND 8	22	1,000	2003	175–500[d]	86 ± 24	64	26	9	0	0	0	2	580	144
	22	1,000	2009	175–500[d]	89 ± 24	64	26	9	0	0	0	2	565	139
AND 9	16	1,000	2003	175–500[d]	84 ± 25	92	4	0	0	0	0	4	630	434
	16	1,000	2009	175–500[d]	85 ± 25	92	5	0	0	0	0	3	602	417
COY 2[b]	--[f]	--[f]	2011	150–350[g]	89	61	0	0	10	17	11	1	1,169	172
COY 4[b]	--[f]	--[f]	2011	150–350[g]	66	55	5	0	18	11	5	6	975	183

Basal area is mean ± standard deviation. PSME = *Pseudotsuga menziesii* (Douglas-fir); TSHE = *Tsuga heterophylla* (western hemlock); THPL = *Thuja plicata* (western red cedar); ABCO = *Abies concolor* (white fir); CADE = *Calocedrus decurrens* (California incense cedar); PILA = *Pinus lambertiana* (sugar pine); -- = not available.

[a]Other (at Coyote Creek) includes *Arbutus menziesii* (madrone), *Pinus ponderosa* (ponderosa pine), and *Taxus brevifolia* (Pacific yew). Other (at the Andrews Forest) includes *Acer macrophyllum* (bigleaf maple), *Castanopsis chrysophylla* (giant chinquapin), and *Prunus emarginata* (bitter cherry).

[b]Based on 2011 stand exam data for matrix (not forest plantations) from Anderson et al., 2013.

[c]Source: Arthur, 2007.

[d]Multi-age stand with mixed-severity fire history.

[e]Coyote 1 was sampled in 2006 (Arthur, 2007) and 2011 (Anderson et al., 2013).

[f]Data from a forestry stand examination, not from plots, and no standard error is provided.

[g]Source: Rothacher, 1969.

Basal area and growth rates in the 34- to 43-year-old plantations in the treated basins are at the lower end of those reported for managed plantations in the region (Figure 3). Basal area at the most recent measurement period (2007 to 2010) ranged from 27 to 35 m^2/ha, or between one third and one half of the basal area in the corresponding reference basin (Table 2). Douglas-fir, which was planted in the treated basins, was the dominant species, representing more than 80% of basal area. Stem density was 5 to 10 times higher in plantations than matched reference basins and ranged from 533 to more than 1,700 stems per hectare (Table 2). Mean diameters in plantations were one third to one fifth of those in corresponding reference basins, except for COY 1, where the large mean stem diameter (31 cm) reflects the retention of 50% of the overstory from the shelterwood harvest (Tables 1 and 2). Trees were smallest in AND 7 (shelterwood harvest, plantation aged 34 years) and largest in 100% clearcut and burned basins AND 1 (plantation, aged 40 years) and COY 4 (plantation, aged 35 years). AND 10, which was clearcut but not burned, had a high number of small stems (plantation, aged 35 years; Tables 1 and 2; Figure 2). Adjusting for age, rates of basal area growth were similar in all the 100% clearcut basins. The unburned basin (AND 10) and the shelterwood harvest basin (AND 7) had slightly lower rates of growth in the third decade after harvest (AND 10) and a precommercial thin (12% basal area removal) at year 28 in AND 7, but rates were similar by 35 years (Figure 3).

The daily soil water balance for the reference basin (AND 2, Figure 4) reveals extremely low rates of evapotranspiration and soil moisture in old-growth forests during the summer (July through September). Evapotranspiration is limited by low temperature in winter and low soil moisture in summer.

Daily streamflow has not changed in reference basins (Figure 5). Runoff declined slightly during the periods of snowmelt, but these minor changes were significant only at AND 2 (Figure 5). Summer streamflow did not change over time.

Conversion of old-growth forest to Douglas-fir plantations, which reached 34 to 43 years of age by the end of the record analyzed here,

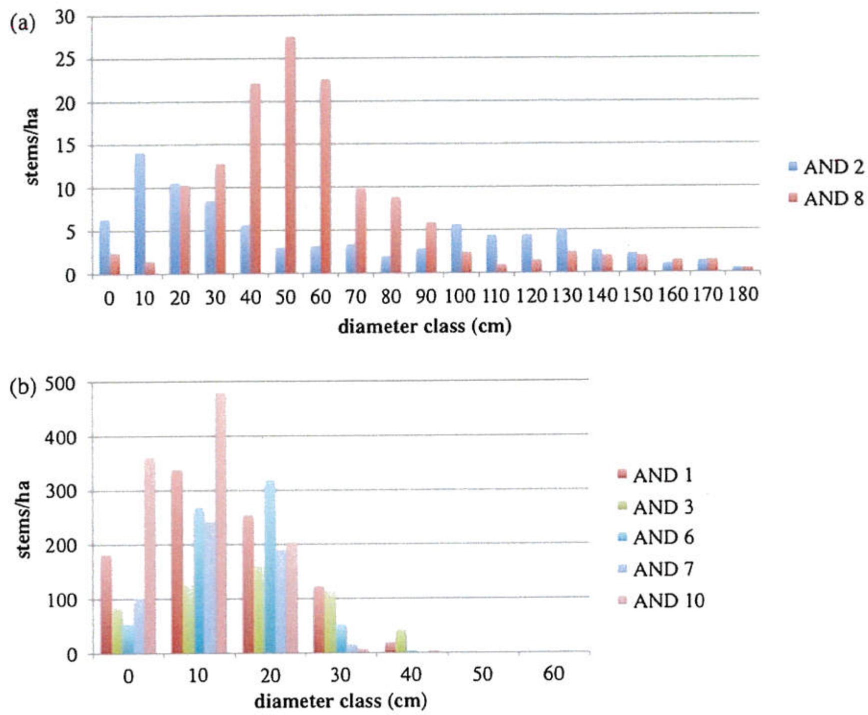


FIGURE 2 Size class distributions of Douglas-fir (*Pseudotsuga menziesii*, PSME) in plantations and reference basins in the Andrews Forest. (a) Reference basins used in this study: AND 2 (2006), AND 8 (2009). (b) Basins with young Douglas-fir plantations: AND 1 (aged 40 years, 2007), AND 3 (clearcut patches, aged 43 years, 2007), AND 6 (aged 34 years, 2008), AND 7 (aged 34 years, 2008), AND 10 (aged 35 years, 2010)

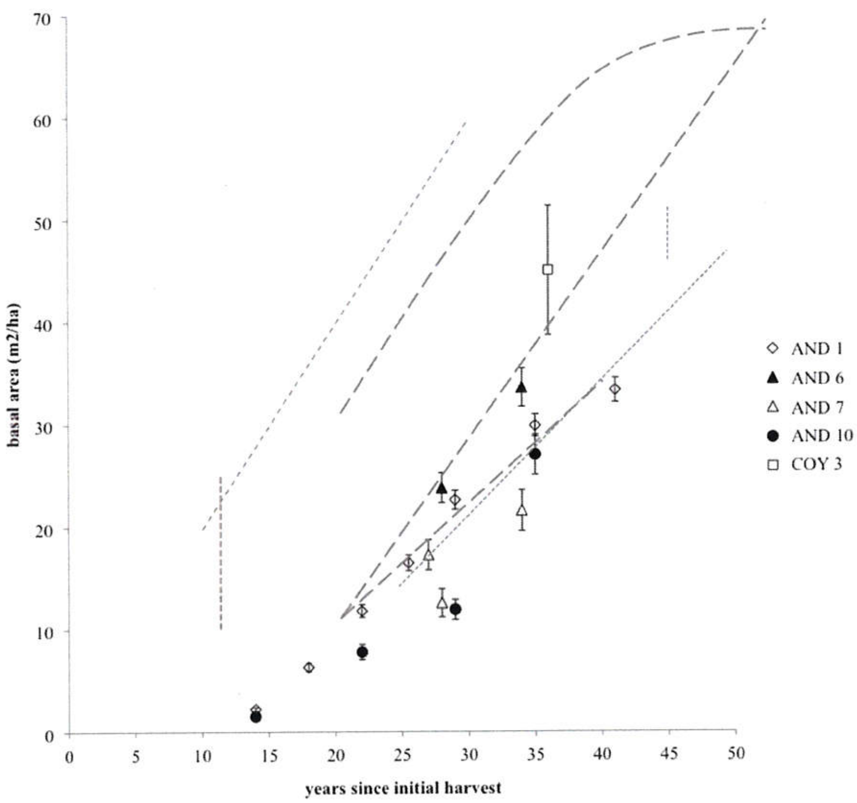


FIGURE 3 Basal area as a function of time since treatment in basins with forest plantations. Symbols are means ± standard error from numbers of plots shown in Table 2. The diagonal thick grey dashed lines are the basal area reported from control (unthinned) plots (upper line), heavily thinned plots (lower line), and lightly thinned plots (middle line) in the Hoskins levels-of-growing-stock (LOGS) installation (site II) in western Oregon (Marshall & Curtis, 2002). The diagonal thin grey dashed line indicates average annual basal area for Douglas-fir plantations on relatively high site productivity locations affected by various levels of infection from Swiss needle cast in the Oregon Coast Range (Maguire, Kanaskie, Voelker, Johnson, & Johnson, 2002). The thin grey diagonal dotted line indicates basal areas for experimental Douglas-fir plantations at low site productivity locations (site V) at Wind River (100 km north of the Andrews Forest, at a similar elevation to the experimental basins; Harrington & Reukema, 1983). The vertical grey dotted line is estimated Douglas-fir basal area from growth and yield models for 45-year-old stands (Marshall & Turnblom, 2005). The vertical grey dashed line is range of basal areas in stands of Douglas-fir, western hemlock, and mixtures (Amoroso & Turnblom, 2006)

had a major effect on summer streamflow. By the mid-1990s, average daily flow in summer (June through September) in basins with plantation forests had declined by roughly 50% relative to the reference basins with 150- to 500-year-old forests (Figure 6a). When plotted by time since harvest, summer streamflow deficits appeared when plantation forests reached 15 years of age (Figure 6b). The trend of declining summer streamflow was temporarily reversed in the late 1980s, especially at AND 1/2 and AND 6/8, after a severe freezing

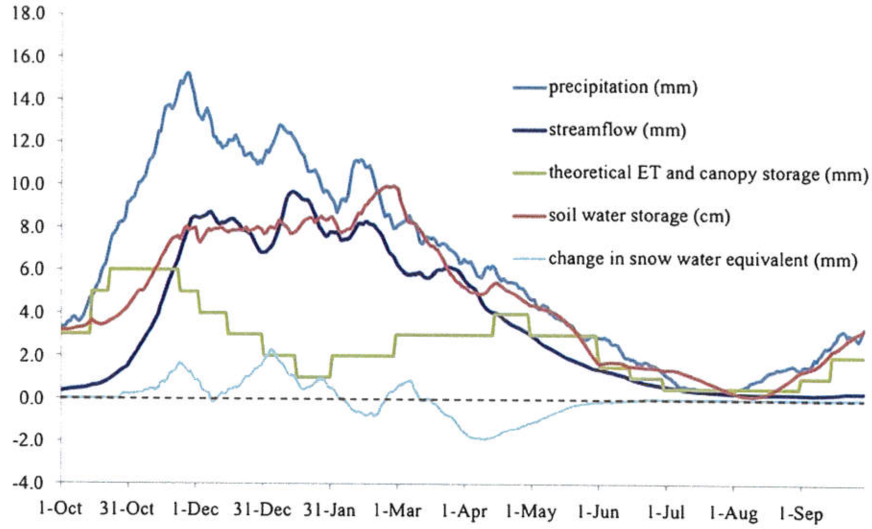


FIGURE 4 Water balance of mean daily values of precipitation (P), streamflow (Q), ET, snow water equivalent (N), and soil water storage (S) in AND 2, based on data from 1953 to 2003 water years, where S = P − Q − ET − ΔN. Daily ET was estimated from the response of AND 1/2 to clearcutting calculated by Jones and Post (2004) and from summer sapflow measured in AND 2 by Moore et al., (2004). Snow water equivalent was based on average modeled daily values from Perkins and Jones (2008)

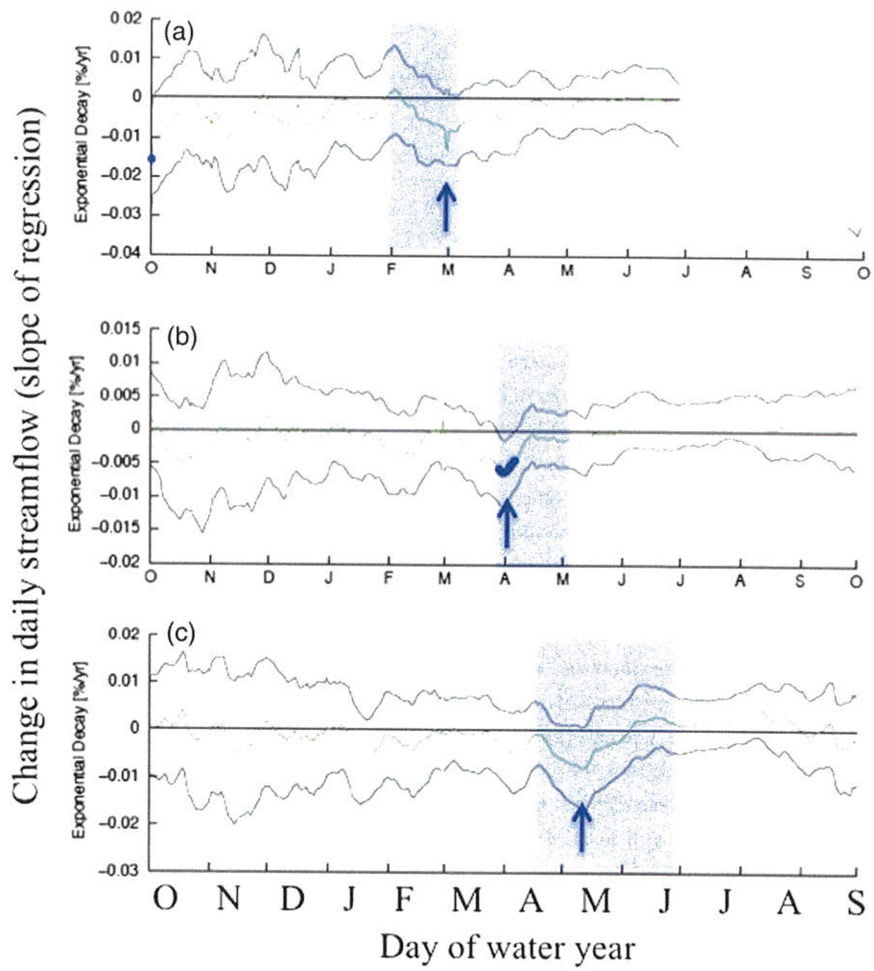


FIGURE 5 Streamflow change for period of record to 1996, by day of water year (October to September) for three reference basins: (a) AND 9 (400 to 700 m), (b) AND 2 (500 to 1,000 m), and (c) AND 8 (800 to 1,100 m). The green line is the trend in streamflow (positive or negative) on that day of the year, relative to the long-term mean streamflow on that day (indicated as zero). Black lines are the 95% confidence interval around the trend. Blue arrows indicate days of declining streamflow, and dark blue lines are days of significant declines in streamflow; declines are significant only at AND 2. Shaded boxes show the period of snowmelt from Perkins and Jones (2008). K. Moore, unpublished data

event in November of 1986. A pre-commercial thin (12% basal area) in AND 7 in 2001 did not slow the decline of summer streamflow.

When examined by day of year, forest harvest produced large streamflow increases from June through December in the first 10 years after harvest (Figure 7). Initial summer streamflow surpluses were lowest, and disappeared most quickly, in 50% thinned ("shelterwood") basins (AND 7, COY 1), and they were highest at the 100% clearcut basins (AND 1, 6, 10, COY 3; Figure 7). Conversion of mature and old forest to young plantations produced streamflow surpluses in winter and spring of 25% to 50%, which persisted virtually unchanged to the present in the Andrews Forest, but not at the drier, more southerly Coyote Creek (Figure 7).

By 20 to 25 years after 100% clearcutting, summer streamflow was lower in all plantation forests compared to reference basins

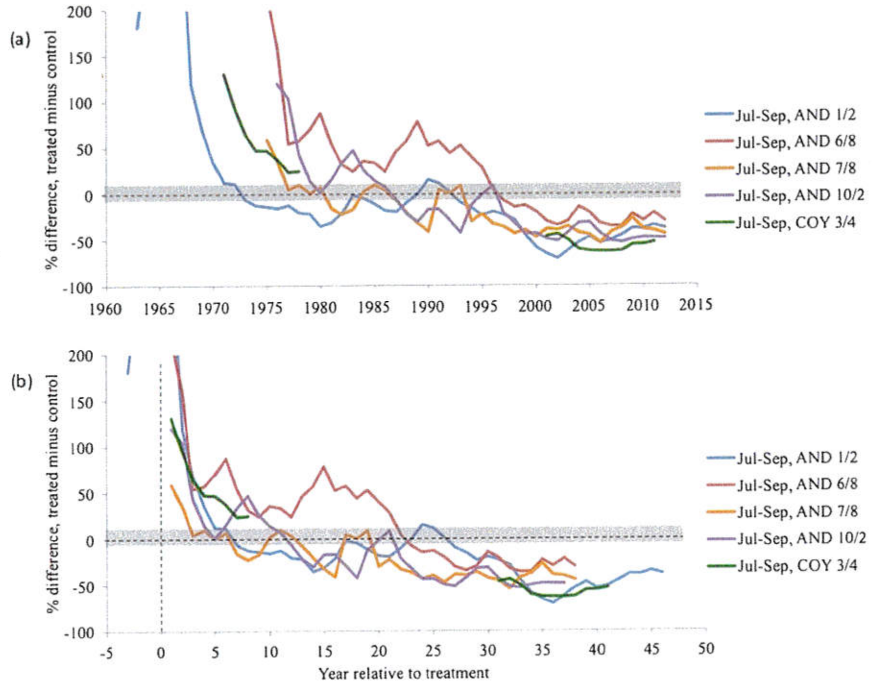


FIGURE 6 Trends in average daily streamflow (July through September) in basins with forest plantations as a percent of streamflow in the reference basin, for five basin pairs with 100% clearcut basins. (a) by year and (b) by time since treatment. Basin pair names include treated/reference. Percents are 3-year running means. Grey box is the mean ± the standard error of the treated-reference basin relationship from July to September during the pre-treatment period. Vertical axis maximum omits years when summer streamflow (July through September) at the treated basin exceeded 200% of pre-treatment level. Maximum percent increases (in unsmoothed data) were 683% at AND 1 (in 1966, fourth year of 1962-1966 clearcutting treatment); 328% at AND 6 (in 1975, one year after treatment); 90% at AND 7 (in 1974, year of treatment); 203% at AND 10 (in 1976, one year after treatment); and 149% at COY 3 (in 1971, year of treatment). Blue detached line shows initial increase when clearcutting (1962 to 1966) began in AND 1. Blue line shows apparent "hydrologic recovery" at AND 1 circa 1990 noted by Hicks et al. (1991); while red line shows increasing streamflow after 1986; both trends are attributable to an extreme freezing event that killed regenerating vegetation. Overall pattern shows no hydrologic recovery to pre-treatment conditions

(Figure 7a–e) and also in one 25% patch cut basin (Figure 7g). In 100% clearcut basins, summer streamflow deficits began by early July, and persisted until early October (AND 1, AND 7, Figure 7a,c), to the end of November (AND 6, AND 10, Figure 7b,d), or to the end of December (COY 3, Figure 7e). Deficits were largest in August and September, when streamflow from forest plantations was 50% lower than from reference basins. Summer deficits did not emerge over time in treatments involving shelterwood (50% thinned, COY 1) and very small openings (0.6- to 1.3-ha patch cuts, COY 2; Figure 7f,h). Relative to 50% thinning (shelterwood) and very small openings, intermediate-sized openings (8-ha patch cuts, AND 3) produced larger initial summer surpluses and persistent summer deficits. The largest openings (20- to 100-ha clearcuts) produced the largest summer surpluses and the largest, persistent summer deficits, which extended into the fall season (Figure 7a–d). Thinning of young forest (AND 7) did not counteract summer streamflow deficits.

Summer streamflow deficits occurred during the period of minimum flow, when soil moisture is most limiting (Figures 4 and 7). The duration of summer streamflow deficits (defined as the difference in the number of days below the first percentile in basins with plantations vs. reference basins) was greater during dry compared to wet summers, at low compared to high elevation, and at the more southerly Coyote Creek compared to the Andrews Forest (Figure 8). Forest plantations that were aged 25 to 35 years in 1995 to 2005 had as many as 100 more days with flow below the first percentile compared to the reference basin (Figure 8). Within a basin pair, the number of days of flow below the first percentile increased in dry relative to wet summers (Figure 8).

5 | DISCUSSION

This study showed that, relative to mature and old-growth forest dominated by Douglas-fir and western hemlock or mixed conifers, forest plantations of native Douglas-fir produced summer streamflow deficits within 15 years of plantation establishment, and these deficits have persisted and intensified in 50-year-old forest stands. Forest stands in the study basins, which are on public forest land, are representative of managed (including thinned) forest stands on private land in the region, in terms of basal area over time (Figure 3), age (10 to 50 years), clearcut size (20 ha), and average rotation age (50 years) (Lutz & Halpern, 2006; Briggs, 2007). There are no significant trends in annual or summer precipitation (Abatzoglou, Rupp, & Mote, 2014) or streamflow at reference basins over the study period. This finding has profound implications for understanding of the effects of land cover change, climate change, and forest management on water yield and timing in forest landscapes.

The size of canopy opening explained the magnitude and duration of initial summer streamflow surpluses and subsequent streamflow deficits, consistent with work on soil moisture dynamics of canopy gaps. In 1990, Gray, Spies, and Easter (2002) created experimental gaps in mature and old-growth forests in Oregon and Washington, including neighboring sites to the study basins, with gap sizes of 40 to 2,000 m^2 (tree height to gap size ratios of 0.2 to 1.0). The smallest gaps dried out faster during the summer than the largest gaps, with the highest moisture levels in the medium-sized gaps, which had less direct radiation and less vigorous vegetation than the largest gaps. In

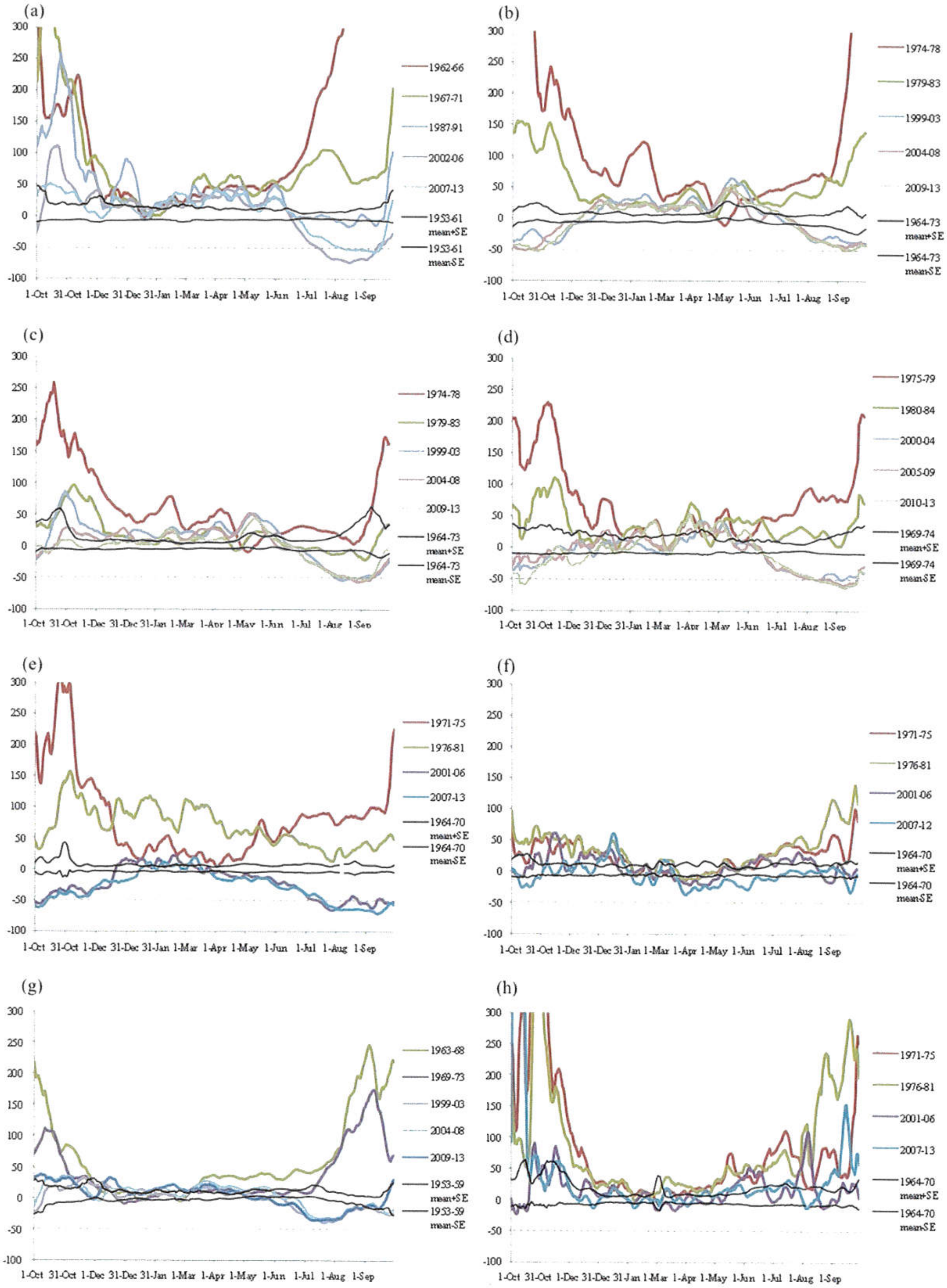


FIGURE 7 Percent change in streamflow by day of water year in 5-year periods after forest harvest and plantation establishment for eight pairs of basins. (a) AND 1 (100% clearcut 1962–66) versus AND 2 (reference), (b) AND 6 (100% clearcut 1974) versus AND 8 (reference), (c) AND 7 (50% cut 1974, remainder cut 1984) versus AND 8 (reference), (d) AND 10 (100% clearcut 1975) versus AND 2 (reference), (e) COY 3 (100% clearcut 1970) versus COY 4 (reference), (f) COY 1 (50% cut 1970) versus COY 4 (reference), (g) AND 3 (25% patch cut 1963) versus AND 2 (reference), (h) COY 2 (30% patch cut 1970) versus COY 4 (reference). Black lines represent the mean and standard error of the percent difference between the treated and reference basins during the pretreatment period. Dashed grey line is a 50% decline in streamflow at the treated basin relative to its relationship to the reference basin during the pretreatment period

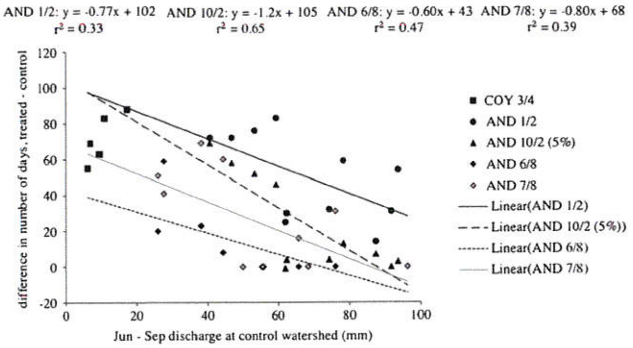


FIGURE 8 Difference in number of days in the first and fifth (AND 10/2) flow percentiles from 1995 to 2005, in basins with 25- to 40-year-old plantations relative to reference (old growth) basins. A value of 0 on the Y-axis indicates that the basin with forest plantation had the same number of days in the low flow percentile as the reference basin; a value of 80 indicates that the basin with forest plantation had 80 more days in the low flow percentile than the reference basin. Negative slopes of regression lines indicate that the duration of low streamflow increased in drier summers in the forest plantation, relative to the reference basin. The fifth percentile was used for AND 10/2 because only a few years had >0 day in the 1% category

late summer (September), volumetric soil moisture declined to 15% in references, 18% in small gaps, and 22% in each of the first 3 years after gap creation (Gray et al., 2002). Together, the paired basin and experimental gap results indicate that even-aged plantations in 8 ha or larger clearcuts are likely to develop summer streamflow deficits, and these deficits are unlikely to be substantially mitigated by dispersed thinning or small gap creation.

Relatively high rates of summer evapotranspiration by young (25 to 45 years old) Douglas-fir plantations relative to mature and old-growth forests apparently caused reduced summer streamflow in treated basins. Young Douglas-fir trees (in AND 1) had higher sapflow per unit sapwood area and greater sapwood area compared to old Douglas-fir trees (in AND 2; Moore, Bond, Jones, Phillips, & Meinzer, 2004). In summer, young Douglas-fir trees have higher rates of transpiration (sapflow) compared to old Douglas-fir trees, because their fast growth requires high sapwood area and because their needles appear to exercise less stomatal control when vapor pressure deficits are high. Leaf area is concentrated in a relatively narrow height range in the forest canopy of a forest plantation, whereas leaf area is distributed over a wide range of heights in a mature or old-growth conifer forest. In summer, these factors appear to contribute to higher daily transpiration rates by young conifers relative to mature or older conifers, producing pronounced reductions in streamflow during the afternoons of hot dry days (Bond et al., 2002). At sunset, transpiration ceases, and streamflow recovers. Hence, daily transpiration produces large diel variations in streamflow in AND 1 (plantation) relative to AND 2 (reference). Other factors, such as differences in tree species composition (Table 2), the presence of a hyporheic zone, or deciduous trees in the riparian zone of AND 1, may also contribute to differences in streamflow between these basins (Bond et al., 2002; Moore et al., 2004; Wondzell, Gooseff, & McGlynn, 2007).

Reduced summer streamflow has potentially significant effects on aquatic ecosystems. Summer streamflow deficits in headwater basins may be particularly detrimental to anadromous fish, including steelhead and salmon, by limiting habitat, exacerbating stream temperature warming, and potentially causing large-scale die-offs (Hicks et al., 1991; Arismendi, Johnson, Dunham, Haggerty, & Hockman-Wert, 2012; Arismendi, Safeeq, Johnson, Dunham, & Haggerty, 2013; Isaak, Wollrab, Horan, & Chandler, 2012). Summer streamflow deficits may also exacerbate trade-offs in water use between in-stream flows, irrigation, and municipal water use.

Reductions in summer streamflow in headwater basins with forest plantations may affect water yield in much larger basins. Much of the Pacific Northwest forest has experienced conversion of mature and old-growth forests to Douglas-fir plantations over the past century. Climate warming and associated loss of snowpack is expected to reduce summer streamflow in the region (e.g., Littell et al., 2010). Declining summer streamflows in the Columbia River basin may be attributed to climate change (Chang, Jung, Steele, & Gannett, 2012; Chang et al., 2013; Hatcher & Jones, 2013), but these declines may also be the result of cumulative forest change due to plantation establishment, fire suppression (Perry et al., 2011), and forest succession after wildfire and insect outbreaks, which kill old trees and promote growth of young forests (e.g., Biederman et al., 2015).

Air temperature has warmed slightly in the Pacific Northwest (0.6 to 0.8°C from 1901 to 2012; Abatzoglou et al., 2014), but water yields from mature and old-growth forests in reference basins have not changed over time. In the reference basins used in this study, we observed small changes in biomass and shifts in species dominance, consistent with changes expected as part of forest succession in mature and old-growth forests, but we did not observe large-scale mortality documented by van Mantgem et al. (2009).

This study demonstrates that plantations of native tree species produced summer streamflow deficits relative to mature and old-growth forest, consistent with prior studies in the U.S. Pacific Northwest (Jones & Post, 2004) and in mixed-deciduous forests in the eastern United States (Hornbeck, Martin, & Eagar, 1997). Research is needed to compare these effects to declining water yield from plantations of fast-growing non-native species in the southern hemisphere (Little et al., 2009; Little, Cuevas, Lara, Pino, & Schoenholtz, 2014; Scott, 2005; Farley et al., 2005). Despite summer streamflow deficits, young forest plantations in the Andrews Forest yield more water in winter, contributing to increased flooding (Harr & McCorison, 1979; Jones & Grant, 1996; Beschta, Pyles, Skaugset, & Surfleet, 2000; Jones, 2000; Jones & Perkins, 2010).

6 | CONCLUSIONS

Paired basin experiments are central to advancing long-term, integrated forest hydrology. Over the past half-century, many key paired-basin experiments (e.g., at U.S. Forest Service Experimental Forests and LTER sites such as Coweeta, Hubbard Brook, and Andrews) have evolved into headwater ecosystem studies, with detailed information about hydrology, climate, vegetation, biogeochemistry, and sediment export. These studies provide rigorous causal inferences about effects of changing vegetation on streamflow at successional time scales (multiple decades) of interest in basic ecology, applied forestry, and conservation. They permit researchers to distinguish forest management from climate change effects on streamflow. Paired-basin experiments are place-based science, integrate multiple disciplines of science and policy, and can dispel assumptions and conjectures such as equilibrium, common in hydrological modeling studies.

Long-term paired-basin studies extending over six decades revealed that the conversion of mature and old-growth conifer forests to plantations of native Douglas-fir produced persistent summer streamflow deficits of 50% relative to reference basins, in plantations aged 25 to 45 years. This result challenges the widespread assumption of rapid "hydrologic recovery" following forest disturbance. Widespread transformation of mature and old-growth forests may contribute to summer water yield declines over large basins and regions around the world, reducing stream habitats and sharpening conflict over uses of water.

Continued research is needed to examine how forest management influences streamflow deficits. Comparative studies, process studies, and modeling are needed to examine legacies of various past and present forestry treatments and effects of native versus non-native tree species on streamflow. In addition, long-term basin studies should be maintained, revived, and extended to a variety of forest types and forest ownerships, in order to discriminate effects of climate versus forest management on water yield and timing, which will be increasingly important in the future.

ACKNOWLEDGMENTS

This study was supported by funding to the Andrews Forest from the NSF Long-term Ecological Research (LTER) program (NSF 1440409, NSF 0823380) and U.S. Forest Service Pacific Northwest Research Station support of hydrology and climate records at the H.J. Andrews Experimental Forest and South Umpqua Experimental Forest. We would like to acknowledge the originators of these studies (J. Rothacher, R. Fredriksen, C.T. Dyrness). We thank Paul Anderson, Craig Creel, Greg Downing, Sean Fleming, John Hammond, Don Henshaw, Mikeal Jones, Rob Pabst, Klaus Puettmann, Suzanne Remillard, Fred Swanson, and USFS staff at Tiller Ranger District of the Umpqua National Forest and the McKenzie River Ranger District of the Willamette National Forest for assistance with data collection, analysis, and interpretation. The manuscript benefitted from comments from two anonymous reviewers.

REFERENCES

Abatzoglou, J. T., Rupp, D. E., & Mote, P. W. (2014). Seasonal climate variability and change in the Pacific Northwest of the United States. *Journal of Climate*, *27*, 2125–2142.

Adam, J. C., Hamlet, A. F., & Lettenmaier, D. P. (2009). Implications of global climate change for snowmelt hydrology in the twenty-first century. *Hydrological Processes*, *23*, 962–972.

Amoroso, M. M., & Turnblom, E. C. (2006). Comparing productivity of pure and mixed Douglas-fir and western hemlock plantations in the Pacific Northwest. *Canadian Journal of Forest Research*, *36*, 1484–1496.

Anderson, P., Rusk, A., Jones, M., Harris, M., Owens, D., Huffman, E. 2013. Coyote Creek research prospectus including 2011 stand exam data. Unpublished report. Tiller Ranger District, Umpqua National Forest.

Andréassian, V. (2004). Waters and forests: From historical controversy to scientific debate. *Journal of Hydrology*, *291*, 1–27.

Arismendi, I., Johnson, S. L., Dunham, J. B., Haggerty, R., & Hockman-Wert, D. (2012). The paradox of cooling streams in a warming world: Regional climate trends do not parallel variable local trends in stream temperature in the Pacific continental United States. *Geophysical Research Letters*, *39*. DOI: 10.1029/2012GL051448.L10401

Arismendi, I., Safeeq, M., Johnson, S. L., Dunham, J. B., & Haggerty, R. (2013). Increasing synchrony of high temperature and low flow in western North American streams: Double trouble for coldwater biota? *Hydrobiologia*, *712*, 61–70.

Arthur, A. S. (2007). Thirty-five years of forest succession in southwest Oregon: Vegetation response to three distinct logging treatments. MS thesis, Oregon State University.

Barnett, T. P., Pierce, D. W., Hidalgo, H. G., Bonfils, C., Santer, B. D., Das, T., ... Cayan, D. R. (2008). Human-induced changes in the hydrology of the western United States. *Science*, *319*, 1080–1083.

Beschta, R. L., Pyles, M. R., Skaugset, A. E., & Surfleet, C. G. (2000). Peakflow responses to forest practices in the western cascades of Oregon, USA. *Journal of Hydrology*, *233*, 102–120.

Biederman, J. A., Somor, A. J., Harpold, A. A., Gutmann, E. D., Breshears, D. D., & Troch, P. A. (2015). Recent tree die-off has little effect on streamflow in contrast to expected increases from historical studies. *Water Resources Research*, *51*. DOI: 10.1002/2015WR017401

Bond, B. J., Jones, J. A., Moore, G., Phillips, N., Post, D., & McDonnell, J. J. (2002). The zone of vegetation influence on baseflow revealed by diel patterns of streamflow and vegetation water use in a headwater basin. *Hydrological Processes*, *16*, 1671–1677.

Briggs, D. (2007). Management practices on Pacific Northwest West-side industrial forest lands, 1991–2005: With projections to 2010. Stand Management Cooperative Working Paper No. 6, College of Forest Resources, University of Washington, Seattle.

Brown, A. E., Zhang, L., McMahon, T. A., Western, A. W., & Vertessy, R. A. (2005). A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. *Journal of Hydrology*, *310*, 28–61.

Budyko, M. I. (1974). Climate and Life, 508 pp. *Academic, San Diego, Calif*, pp.72–191.

Chang, H., Jung, I. W., Steele, M., & Gannett, M. (2012). Spatial patterns of March and September streamflow trends in Pacific Northwest streams, 1958–2008. *Geographical Analysis*, *44*, 177–201.

Chang, H., Jung, I. W., Strecker, A., Wise, D., Lafrenz, M., & Shandas, V. (2013). Water supply, demand, and quality indicators for assessing the spatial distribution of water resource vulnerability in the Columbia River basin. *Atmosphere-Ocean*, *51*, 339–356.

Cornish, P. M., & Vertessy, R. A. (2001). Forest age-induced changes in evapotranspiration and water yield in a eucalypt forest. *Journal of Hydrology*, *242*, 43–63.

Creed, I. F., Spargo, A. T., Jones, J. A., Buttle, J. M., Adams, M. B., Beall, F. D., et al. (2014). Changing forest water yields in response to climate warming: Results from long-term experimental watershed sites across North America. *Global Change Biology*, *20*, 3191–3208.

Dai, A., Qian, T., Trenberth, K. E., & Milliman, J. D. (2009). Changes in continental freshwater discharge from 1948 to 2004. *Journal of Climate*, *22*, 2773–2792.

Dyrness, C. T. (1967). Mass soil movements in the H.J. Andrews Experimental Forest. Res. Pap. PNW-42. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station (p. 13).

Dyrness, C. T. (1969). Hydrologic properties of soils on three small watersheds in the western Cascades of Oregon. Res. Note PNW-111. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station (p. 17).

Dyrness, C. T., & Hawk, G. (1972). Vegetation and soils of the Hi-15 watersheds, H.J. Andrews Experimental Forest. Seattle: University of Washington; Coniferous For. Biome Internal Rep. 43 (p. 28).

Eberhardt, L. L., & Thomas, J. M. (1991). Designing environmental field studies. *Ecological Monographs*, *61*, 53–73.

Farley, K. A., Jobbágy, E. G., & Jackson, R. B. (2005). Effects of afforestation on water yield: A global synthesis with implications for policy. *Global Change Biology*, *11*, 1565–1576.

Gray, A. N., Spies, T. A., & Easter, M. J. (2002). Microclimatic and soil moisture responses to gap formation in coastal Douglas-fir forests. *Canadian Journal of Forest Research*, *32*, 332–343.

Halpern, C. B. (1989). Early successional patterns of forest species: Interactions of life history traits and disturbance. *Ecology*, *70*, 704–720.

Halpern, C. B., & Franklin, J. F. (1990). Physiognomic development of Pseudotsuga forests in relation to initial structure and disturbance intensity. *Journal of Vegetation Science*, *1*, 475–482.

Halpern, C. B., & Lutz, J. A. (2013). Canopy closure exerts weak controls on understory dynamics: A 30-year study of overstory–understory interactions. *Ecological Monographs*, *83*, 221–237.

Halpern, C. B., & Spies, T. A. (1995). Plant species diversity in natural and managed forests of the Pacific Northwest. *Ecological Applications*, *5*, 913–934.

Hamlet, A. F., & Lettenmaier, D. P. (2007). Effects of 20th century warming and climate variability on flood risk in the western US. *Water Resources Research*, *43*. DOI: 10.1029/2006WR005099.W06427

Harr, R. D., Fredriksen, R. L., & Rothacher, J. (1979). Changes in streamflow following timber harvest in southwestern Oregon. Res. Pap. PNW-249. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station (p. 22).

Harr, R. D. (1981). Some characteristics and consequences of snowmelt during rainfall in western Oregon. *Journal of Hydrology*, *53*, 277–304.

Harr, R. D. (1986). Effects of clearcutting on rain-on-snow runoff in western Oregon: A new look at old studies. *Water Resources Research*, *22*, 1095–1100.

Harr, R. D., & McCorison, F. M. (1979). Initial effects of clearcut logging on size and timing of peak flows in a small watershed in western Oregon. *Water Resources Research*, *15*, 90–94.

Harr, R. D., Levno, A., & Mersereau, R. (1982). Streamflow changes after logging 130-year-old Douglas fir in two small watersheds. *Water Resources Research*, *18*, 637–644.

Harrington, C. A., & Reukema, D. L. (1983). Initial shock and long-term stand development following thinning in a Douglas-fir plantation. *Forest Science*, *29*, 33–46.

Hatcher, K. L., & Jones, J. A. (2013). Climate and streamflow trends in the Columbia River basin: Evidence for ecological and engineering resilience to climate change. *Atmosphere–Ocean*, *51*, 436–455.

Hicks, B. J., Beschta, R. L., & Harr, R. D. (1991). Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. *Journal of the American Water Resources Association*, *27*, 217–226.

Hodgkins, G. A., Dudley, R. W., & Huntington, T. G. (2003). Changes in the timing of high river flows in New England over the 20th century. *Journal of Hydrology*, *278*, 244–252.

Hodgkins, G. A., Dudley, R. W., & Huntington, T. G. (2005). Summer low flows in New England during the 20th century. *Journal of the American Water Resources Association*, *41*, 403–412.

Hornbeck, J. W., Martin, C. W., & Eagar, C. (1997). Summary of water yield experiments at Hubbard Brook experimental forest, New Hampshire. *Canadian Journal of Forest Research*, *27*, 2043–2052.

Isaak, D. J., Wollrab, S., Horan, D., & Chandler, G. (2012). Climate change effects on stream and river temperatures across the northwest U.S. from 1980–2009 and implications for salmonid fishes. *Climatic Change*, *113*, 499–524.

Jefferson, A., Nolin, A., Lewis, S., & Tague, C. (2008). Hydrogeologic controls on streamflow sensitivity to climate variation. *Hydrological Processes*, *22*, 4371-4385.

Jennings, K., & Jones, J. A. (2015). Precipitation-snowmelt timing and snowmelt augmentation of large peak flow events, western Cascades, Oregon. *Water Resources Research*, *51*, 7649-7661. DOI:10.1002/2014WR016877.

Jones, J. A. (2000). Hydrologic processes and peak discharge response to forest removal, regrowth, and roads in 10 small experimental basins, western Cascades, Oregon. *Water Resources Research*, *36*, 2621–2642.

Jones, J. A. (2011). Hydrologic responses to climate change: Considering geographic context and alternative hypotheses. *Hydrological Processes*, *25*, 1996–2000.

Jones, J. A., & Grant, G. E. (1996). Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. *Water Resources Research*, *32*, 959–974.

Jones, J. A. & Perkins, R. M. (2010). Extreme flood sensitivity to snow and forest harvest, western Cascades, Oregon, United States. *Water Resources Research*, *46*, W12512. DOI: 10.1029/2009WR008632

Jones, J. A. & Post, D. A. (2004). Seasonal and successional streamflow response to forest cutting and regrowth in the northwest and eastern United States. *Water Resources Research*, *40*, W05203. DOI. 10.1029/2003WR002952

Jones, J. A., Creed, I. F., Hatcher, K. L., Warren, R. J., Adams, M. B., Benson, M. H., ... Williams, M. W. (2012). Ecosystem processes and human influences regulate streamflow response to climate change at long-term ecological research sites. *BioScience*, *62*, 390–404.

Kennedy, A. M., Garen, D. C., & Koch, R. W. (2009). The association between climate teleconnection indices and Upper Klamath seasonal streamflow: Trans-Niño Index. *Hydrological Processes*, *23*, 973–984.

Lara, A., Villalba, R., & Urrutia, R. (2008). A 400-year tree-ring record of the Puelo River summer–fall streamflow in the Valdivian Rainforest ecoregion, Chile. *Climatic Change*, *86*, 331–356.

Lins, H. F., & Slack, J. R. (1999). Streamflow trends in the United States. *Geophysical Research Letters*, *26*, 227–230.

Lins, H. F., & Slack, J. R. (2005). Seasonal and regional characteristics of US streamflow trends in the United States from 1940 to 1999. *Physical Geography*, *26*, 489–501.

Littell, J. S., Oneil, E. E., McKenzie, D., Hicke, J. A., Lutz, J. A., Norheim, R. A., & Elsner, M. M. (2010). Forest ecosystems, disturbance, and climatic change in Washington State, USA. *Climatic Change*, *102*, 129–158.

Little, C., Cuevas, J. G., Lara, A., Pino, M., & Schoenholtz, S. (2014). Buffer effects of streamside native forests on water provision in watersheds dominated by exotic forest plantations. *Ecohydrology*, *8*, 1205–1217.

Little, C., Lara, A., McPhee, J., & Urrutia, R. (2009). Revealing the impact of forest exotic plantations on water yield in large scale watersheds in South-Central Chile. *Journal of Hydrology*, *374*, 162–170.

Luce, C. H., & Holden, Z. A. (2009). Declining annual streamflow distributions in the Pacific Northwest United States, 1948–2006. *Geophysical Research Letters*, *36*. DOI: 10.1029/2009GL039407.L16401

Lutz, J. A., & Halpern, C. B. (2006). Tree mortality during early forest development: A long-term study of rates, causes, and consequences. *Ecological Monographs*, *76*, 257–275.

Maguire, D. A., Kanaskie, A., Voelker, W., Johnson, R., & Johnson, G. (2002). Growth of young Douglas-fir plantations across a gradient in Swiss needle cast severity. *Western Journal of Applied Forestry*, *17*, 86–95.

Marshall, D.D. and Curtis, R.O., 2002. Levels-of-growing-stock cooperative study in Douglas-fir: Report no. 15–Hoskins: 1963–1998. USDA Forest Service Pacific Northwest Research Station Research Paper PNW-RP-537.

Marshall, D. D., & Turnblom, E. C. (2005). Wood productivity of Pacific Northwest Douglas-fir: Estimates from growth-and-yield models. *Journal of Forestry*, *103*, 71–72.

Marshall, J. D., & Waring, R. H. (1984). Conifers and broadleaf species: Stomatal sensitivity differs in western Oregon. *Canadian Journal of Forest Research*, *14*, 905–908.

Marshall, J. D., & Waring, R. H. (1986). Comparison of methods of estimating leaf-area index in old-growth Douglas-fir. *Ecology*, *67*, 975–979.

McCabe, G. J., & Wolock, D. M. (2002). A step increase in streamflow in the conterminous United States. *Geophysical Research Letters*, *29*, 2185. DOI: 10.1029/2002GL015999.

Moore, G. W., Bond, B. J., Jones, J. A., Phillips, N., & Meinzer, F. C. (2004). Structural and compositional controls on transpiration in 40-and 450-year-old riparian forests in western Oregon, USA. *Tree Physiology*, *24*, 481–491.

Mote, P. W., Parson, E. A., Hamlet, A. F., Keeton, W. S., Lettenmaier, D., Mantua, N., ... Snover, A. K. (2003). Preparing for climatic change: The water, salmon, and forests of the Pacific Northwest. *Climatic Change*, *61*, 45–88.

Nolin, A. W., & Daly, C. (2006). Mapping "at risk" snow in the Pacific Northwest. *Journal of Hydrometeorology*, *7*, 1164–1171.

Perkins, R. M., & Jones, J. A. (2008). Climate variability, snow, and physiographic controls on storm hydrographs in small forested basins, western Cascades, Oregon. *Hydrological Processes*, *22*, 4949–4964.

Perry, D. A., Hessburg, P. F., Skinner, C. N., Spies, T. A., Stephens, S. L., Taylor, A. H., ... Riegel, G. (2011). The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. *Forest Ecology and Management*, *262*, 703–717.

Rothacher, J. (1965). Streamflow from small watersheds on the western slope of the Cascade Range of Oregon. *Water Resources Research*, *1*, 125–134.

Rothacher, J. (1969). A study of the effects of timber harvesting on small watersheds in the sugar pine Douglas-fir area of S.W. Oregon. Coyote Creek establishment report. Unpublished report (p. 12)

Rothacher, J. (1970). Increases in water yield following clear-cut logging in the Pacific Northwest. *Water Resources Research*, *6*, 653–658.

Rothacher, J., Dyrness, C. T., & Fredriksen, R. L. (1967). Hydrologic and related characteristics of three small watersheds in the Oregon Cascades. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station (p. 54)

Scott, D. F. (2005). On the hydrology of industrial timber plantations. *Hydrological Processes*, *19*, 4203–4206.

Stewart, I. T., Cayan, D. R., & Dettinger, M. D. (2004). Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. *Climatic Change*, *62*, 217–232.

Stewart, I. T., Cayan, D. R., & Dettinger, M. D. (2005). Changes toward earlier streamflow timing across western North America. *Journal of Climate*, *18*, 1136–1155.

Sun, G., Zhou, G., Zhang, Z., Wei, X., McNulty, S. G., & Vose, J. M. (2006). Potential water yield reduction due to forestation across China. *Journal of Hydrology*, *328*, 548–558.

Swanson, F. J., & James, M. E. (1975). Geomorphic history of the lower Blue River-Lookout Creek area, Western Cascades, Oregon. *Northwest Science*, *49*, 1–11.

Swanson, F. J., & Swanston, D. N. (1977). Complex mass-movement terrains in the western Cascade Range, Oregon. *Reviews in Engineering Geology*, *3*, 113–124.

Tepley, A. J. (2010). Age structure, developmental pathways, and fire regime characterization of Douglas-fir/Western Hemlock Forests in the Central Western Cascades of Oregon. PhD thesis, Oregon State University.

Tepley, A. J., Swanson, F. J., & Spies, T. A. (2013). Fire-mediated pathways of stand development in Douglas-fir/Western Hemlock forests of the Pacific Northwest, USA. *Ecology*, *94*, 1729–1743.

van Dijk, A. I., & Keenan, R. J. (2007). Planted forests and water in perspective. *Forest Ecology and Management*, *251*, 1–9.

Van Mantgem, P. J., Stephenson, N. L., Byrne, J. C., Daniels, L. D., Franklin, J. F., Fulé, P. Z., ... Veblen, T. T. (2009). Widespread increase of tree mortality rates in the western United States. *Science*, *323*, 521–524.

Vörösmarty, C. J., Green, P., Salisbury, J., & Lammers, R. B. (2000). Global water resources: Vulnerability from climate change and population growth. *Science*, *289*, 284–288.

Weisberg, P. J., & Swanson, F. J. (2003). Regional synchroneity in fire regimes of western Oregon and Washington, USA. *Forest Ecology and Management*, *172*, 17–28.

Wondzell, S. M., Gooseff, M. N., & McGlynn, B. L. (2007). Flow velocity and the hydrologic behavior of streams during baseflow. *Geophysical Research Letters*, *34*, L24404. DOI: 10.1029/2007GL031256.

Zhang, L., Dawes, W. R., & Walker, G. R. (2001). Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research*, *37*, 701–708.

SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

How to cite this article: Perry, T. D., and Jones, J. A. (2016), Summer streamflow deficits from regenerating Douglas-fir forest in the Pacific Northwest, USA, *Ecohydrology*, doi: 10.1002/eco.1790

Oregon State University

» Newsroom

High wildfire severity risk seen in young plantation forests

April 27, 2018

CORVALLIS, Ore. – Wildfires show no respect for property lines, but a new analysis of the 2013 Douglas Complex fire in southwestern Oregon concludes that young plantation forests managed by industrial owners experienced higher severity fire than did nearby public forests.

Researchers in the College of Forestry at Oregon State University used satellite imagery and local data to analyze the factors driving differences of severity in the fire, which burned about 50,000 acres north of Grants Pass. Located in the Klamath Mountains ecoregion, the area is dominated by Douglas fir, ponderosa pine and white fir and is a mix of private and federal ownership and state-owned O&C (Oregon & California Railroad) lands.

While daily weather was the most significant driver of fire severity, the researchers found that other factors such as ownership, forest age and topography were also critical. Intensively managed private forestlands tended to burn with greater severity than older state and federal forests. The findings are important because they point to the need for collaboration among landowners, both public and private, to reduce the wildfire risk across the region.

The findings were published in Ecological Applications, a professional journal.

"We leveraged a fire severity metric that integrates fire intensity and tree susceptibility. We demonstrated how industrial forest management increases these aspects of fire risk," said Harold Zald, lead author and a former post-doctoral scientist at Oregon State. Zald, who is now an assistant professor at Humboldt State University in Arcata, California, worked with Christopher Dunn, who is currently a post-doc at OSU.

"Given the configuration of the checkerboard landscape in southwest Oregon, the probability of fire occurrence is likely very similar across ownerships," added Zald.

The report contrasts with other studies that focus on increasing wildfire risk stemming from fuels that accumulate as a result of historical fire suppression. The common perception that recent increases in wildfire risk — particularly in states with large portions of federal lands — results from mismanagement on those lands is not necessarily correct, the researchers added. "Our study clearly demonstrates that the legacy of historical and current forest management practices for timber production contribute more to increased wildfire risk," said Dunn.

While fire risks may be higher on public lands than they were in the past, privately managed plantation forests in the Douglas Complex fire still tended to show greater fire severity than did forests on public lands.

Private industrial forests tend to be dominated by younger trees that have thinner bark and lower crown height, the researchers wrote. Both those characteristics are known to lead to increased fire-induced tree mortality. However, even after accounting for the age of forested stands, industrial forest management resulted in higher fire severity for reasons that are not entirely clear. One explanation may be that industrial forests have much more homogenous forests, both horizontally and vertically, at spatial scales that influence fire behavior, Zald and Dunn wrote.

The researchers called for more research on how fire is transmitted between lands under different ownerships and management regimes. Alternative management strategies are also needed, they wrote, to reduce fire risks within stands and across the landscape.

The research was funded by a grant from the U.S. Bureau of Land Management.

About the OSU College of Forestry: *For a century, the College of Forestry has been a world class center of teaching, learning and research. It offers graduate and undergraduate degree programs in sustaining ecosystems, managing forests and manufacturing wood products; conducts basic and applied research on the nature and use of forests; and operates 14,000 acres of college forests.*

MULTIMEDIA:

Click photos to see a full-size version. Right click and save image to download.

STORY BY:

Nick Houtman, 541-737-0783, nick.houtman@oregonstate.edu

SOURCE:

Harold Zald, 707-826-5484, hsz16@humboldt.edu; Christopher Dunn, 541-737-1194, chris.dunn@oregonstate.edu

Contact Info

Grab a feed of news and stories for your site.