

Forest Structure Affects Soil Mercury Losses in the Presence and Absence of Wildfire

The Faculty of Oregon State University has made this article openly available.
Please share how this access benefits you. Your story matters.

Citation	Homann, P. S., Darbyshire, R. L., Bormann, B. T., & Morrisette, B. A. (2015). Forest Structure Affects Soil Mercury Losses in the Presence and Absence of Wildfire. <i>Environmental Science & Technology</i> , 49(21), 12714-12722. doi:10.1021/acs.est.5b03355
DOI	10.1021/acs.est.5b03355
Publisher	American Chemical Society
Version	Version of Record
Terms of Use	http://cdss.library.oregonstate.edu/sa-termsfuse

Forest Structure Affects Soil Mercury Losses in the Presence and Absence of Wildfire

Peter S. Homann,^{*,†} Robyn L. Darbyshire,[‡] Bernard T. Bormann,[§] and Brett A. Morrisette^{||}

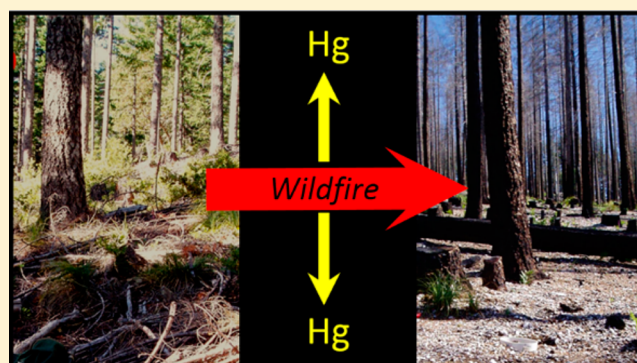
[†]Huxley College of the Environment, Western Washington University, Bellingham, Washington 98225-9181, United States

[‡]Pacific Northwest Region, USDA Forest Service, Portland, Oregon 97204-3440, United States

[§]School of Environmental and Forest Sciences, College of the Environment, University of Washington, Seattle, Washington 98195, United States

^{||}Department of Forest Ecosystems and Society, Oregon State University, Corvallis, Oregon 97331, United States

ABSTRACT: Soil is an important, dynamic component of regional and global mercury (Hg) cycles. This study evaluated how changes in forest soil Hg masses caused by atmospheric deposition and wildfire are affected by forest structure. Pre and postfire soil Hg measurements were made over two decades on replicate experimental units of three prefire forest structures (mature unthinned, mature thinned, clear-cut) in Douglas-fir dominated forest of southwestern Oregon. In the absence of wildfire, O-horizon Hg decreased by 60% during the 14 years after clearcutting, possibly the result of decreased atmospheric deposition due to the smaller-stature vegetative canopy; in contrast, no change was observed in mature unthinned and thinned forest. Wildfire decreased O-horizon Hg by >88% across all forest structures and decreased mineral-soil (0 to 66 mm depth) Hg by 50% in thinned forest and clear-cut. The wildfire-associated soil Hg loss was positively related to the amount of surface fine wood that burned during the fire, the proportion of area that burned at >700 °C, fire severity as indicated by tree mortality, and soil C loss. Loss of soil Hg due to the 200 000 ha wildfire was more than four times the annual atmospheric Hg emissions from human activities in Oregon.



■ INTRODUCTION

Mercury (Hg) is a global pollutant that has potential impacts on human and animal health.¹ It cycles among the atmosphere, oceans, and terrestrial systems.¹ Soils are important, dynamic pools in the mercury cycle. In the U.S.A., forest soil Hg pools are 2 orders of magnitude greater than annual anthropogenic emissions.² Forest soils have the potential to sequester additional Hg or to release extant Hg to the atmosphere and aquatic systems.

The size of the forest soil Hg pool is influenced by multiple factors. Both atmospheric deposition and parent material are sources of soil Hg.² Less atmospheric deposition resulting from a decrease in local and regional anthropogenic Hg emissions is the proposed mechanism of the decrease in soil Hg concentration over a two-decade period in Pennsylvania oak forest.³ Recent multidecadal decreases in anthropogenic mercury emissions across much of North America^{4–6} portend possible decreases in atmospheric deposition to forests and the consequent decreases in soil Hg pools.

Soil Hg may also be affected by vegetation characteristics. Vegetation type can affect soil organic matter, C and N,^{7,8} to which Hg is closely related.^{2,9,10} However, differences in C pools between vegetation types do not necessarily translate into differences in soil Hg pools.¹¹ Vegetation canopy structure

affects atmospheric Hg deposition,^{12–14} with deposition rates being greater in conifer forest than deciduous forest and open areas.¹⁵ The effect of these vegetation-influenced deposition rates on soil Hg pools remains to be determined and is considered in this study.

Forest wildfires redistribute soil Hg to the atmosphere via volatilization and particulate emission¹⁶ and to downslope areas and potentially into waterways via erosion.¹⁷ Forest fires can increase Hg accumulation in fish by altering food webs and by increasing Hg inputs into streams and lakes.¹⁸ In the western U.S.A. and Canada, forest fire extent is forecast to increase,^{19–21} creating concern that enhanced wildfire will release more Hg that will enter foodchains.²²

Wildfire-associated soil Hg losses vary by more than 2 orders of magnitude^{23–27} and may be related to vegetation type and fire severity. Conifer forest lost much more soil Hg than meadow in a Washington state wildfire,²⁶ but high variation among sites prevented similar generalization in a Wyoming wildfire.²⁵ Soil Hg loss was related to fire severity in Wyoming

Received: July 10, 2015

Revised: September 10, 2015

Accepted: October 8, 2015

Published: October 20, 2015

Table 1. Aboveground Structural Characteristics [Mean (se)] Vary among Treatments, 1 to 4 Years Prior to Wildfire in Conifer Forest, Rogue River–Siskiyou National Forest, Oregon^{a,b}

treatment	n expt. units	conifer foliar biomass (kg m ⁻²)	conifer woody biomass (kg m ⁻²)	surface coarse wood (kg m ⁻²)	surface fine wood (kg m ⁻²)
unthinned	5	1.06 a (0.12)	31.9 a (3.3)	1.05 (0.22)	1.27 a (0.25)
thinned	6	0.50 b (0.07)	14.6 b (1.9)	2.45 (0.60)	2.99 b (0.53)
clear-cut	10	0.00 c (0.00)	0.0 c (0.0)	3.74 (1.05)	4.87 c (0.36)

^aAdapted from Homann et al.³⁰ and Bormann et al.³⁸ ^bWithin a column, when differences among means occurred, means not followed by a same letter are different at $P < 0.05$, based on ANOVA followed by Tukey multiple-comparison test.

wildfire,²⁵ and Woodruff and Cannon²⁸ suggested that high fire severity resulted in significant Hg loss from mineral soils as well as organic soils. But, soil Hg losses do not always vary with fire severity.^{26,27}

Forest structure has various definitions, but key structural attributes include tree species, height, diameter, spacing, and biomass; canopy cover and foliage arrangement; understory vegetation; and standing and fallen dead woody debris.²⁹ We expand these concepts to encompass a biogeochemical perspective that considers the amount and distribution of the organic matter and elements within an ecosystem, including the soil. In Minnesota pine forest, disturbance altered forest structure; specifically, prefire blowdown redistributed trees and substantially increased soil Hg mass, leading to greater wildfire-associated soil Hg losses.²⁷ Other prefire disturbances may affect only aboveground forest structure and not alter prefire soils;³⁰ their influence on soil Hg losses is unknown and is addressed in this study.

The purpose of this study was to evaluate the effect of conifer forest structure on soil Hg pools in the presence and absence of wildfire. The following questions were addressed: (i) How does forest structure influence soil Hg pools in the absence of wildfire? (ii) How does forest structure influence soil Hg pools in the presence of wildfire? (iii) How does fire severity influence wildfire-associated soil Hg losses? These questions were addressed by evaluating replicate unburned and burned experimental units for three prefire forest structures: mature unthinned, mature thinned, and clear-cut in a Douglas-fir dominated forest.³⁰ Soil measurements made repeatedly over a two decade period allowed comparison of pre and postwildfire values to quantify changes in soil Hg pools.

METHODS

Study Site and Design. The study site and design have been described in detail in Homann et al.³⁰ and are summarized below. The site is located 25 km southeast of Gold Beach, Oregon, U.S.A., at an elevation between 750 and 900 m in the Rogue River–Siskiyou National Forest. When the study site was established in 1992, the native forest was dominated by 80- to 110-year-old Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco) that had regenerated naturally after stand-replacing wildfire in 1881.³¹ Mean January temperature is 4 °C, mean July temperature is 18 °C, and annual precipitation is approximately 190 cm, of which only 10 cm falls between June and September.³¹ Additional site details are available in Little et al.³¹ and Raymond and Peterson.³²

Prior to wildfire, two forest management experiments were established, encompassing 27 experimental units (6 to 8 ha each) distributed across five mountain slopes within a 5-km radius. Experimental treatments were imposed in 1997 and consisted of clear-cut, thinned, and unthinned (nontreated). The 2002 Biscuit Complex Wildfire and associated backburns burned portions of both experiments as a surface fire,³² thereby

disrupting the original experimental designs. As a consequence, 21 postwildfire experimental units are categorized by treatment and burn status: 2 unthinned unburned, 3 unthinned wildfire-burned, 3 thinned unburned, 3 thinned wildfire-burned, 4 clear-cut unburned, and 6 clear-cut wildfire-burned. Four additional unburned experimental units that were not measured in 2003 and two other units that underwent prescribed burning were not considered in this wildfire analysis. During the period from 1998 to 2000, the total basal area varied among the unthinned experimental units between 46 and 72 m² ha⁻¹, conifer basal area between 37 and 68 m² ha⁻¹, conifer density between 411 and 788 stems ha⁻¹, conifer quadratic mean diameter between 27 and 42 cm, and conifer woody biomass between 20 and 39 kg m⁻². Thinned and clear-cut units had substantially less biomass (Table 1).

Measurements and Data Synthesis. For each experimental unit at each of three sampling times, an experimental-unit value was determined for each Hg pool (mass m⁻²) based on multiple samples and measurements taken within the unit, as described below. The experimental-unit values were then subjected to statistical analysis.

Soil sampling was conducted three times (prefire in 1992–1995, 1-year postfire in 2003, and 9 years postfire in 2011) within a central 1.5-ha sampling area in each experimental unit. Soil samples at an average of 13 points per experimental unit were collected as described in detail by Bormann et al.³³ and Homann et al.^{30,34–36} Sampling points in different years were approximately 2 m apart. O layers were collected from within rings of cross-sectional area 707 cm² in 1992–1995 and 352 cm² in 2003 and 2011. After removing the O layer, the mineral soil was collected volumetrically with a corer of 140 cm² cross-sectional area. In 1992–1995, three mineral-soil layers were sampled: 0 cm to bottom of A horizon, bottom of A to 15 cm, and 15 to 30 cm. In 2003, three layers were sampled: 0–3 cm, 3–15 cm, and 15–30 cm. In 2011, two layers were sampled: 0–3 cm, 3–6 cm. Samples were air-dried or oven-dried (60 °C). The O-layer samples were hand-sorted into >4 mm rocks, >6.4 mm diameter woody debris, and remaining material, which we refer to as soil. Mineral soil samples were sieved at 4 mm; we refer to the <4 mm-fraction as soil. For each sampling time, experimental unit and soil layer, a mass-weighted composite of the soil was made from the sampling points. Samples were stored in low-density polyethylene plastic bags. O-layer subsamples were ground to <1 mm and mineral soil subsamples to <0.5 mm, stored in paper envelopes, and dried at 70 °C prior to chemical analysis. This temperature is very unlikely to substantially affect the total Hg concentrations; for example, in an evaluation of soil from an uncontaminated site, the total Hg concentration of two O horizons, and A, B, and C mineral horizons dried at 105 °C averaged 102% of the corresponding freeze-dried values.³⁷ All composite subsamples were analyzed for Hg during December 2014 and January 2015; subsamples from the different sampling periods were alternated

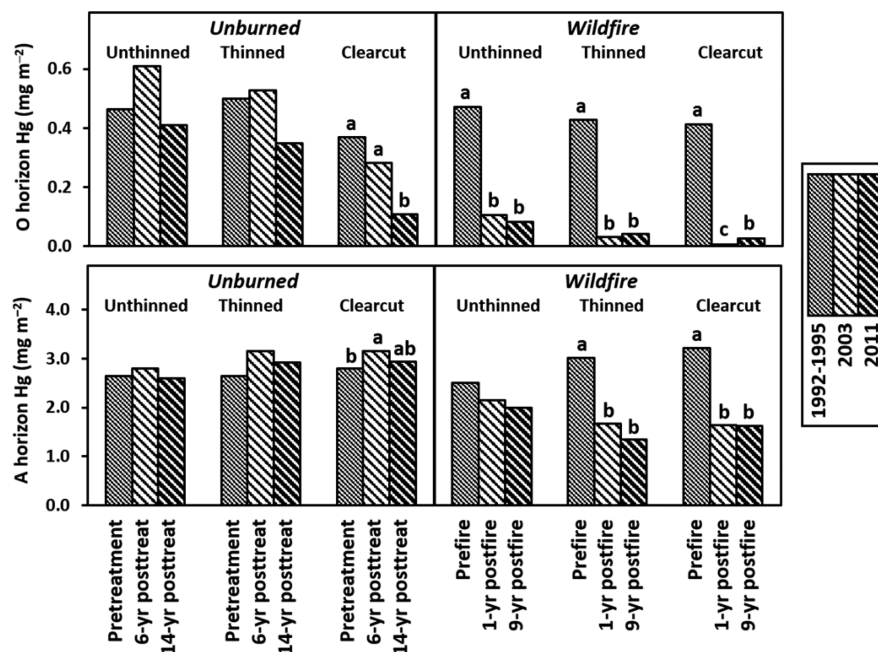


Figure 1. Nineteen-year trajectory of mean O and A-horizon soil Hg masses in conifer forest exposed to different treatments (unthinned, thinned, clear-cut) followed by burn conditions (unburned, wildfire), Rogue River–Siskiyou National Forest, Oregon. Pretreatment–prefire values were taken in 1992–1995; treatments were imposed in 1997; wildfire occurred in 2002; posttreatment–postfire measurements were taken in 2003 and 2011. Within each horizon–burn type–treatment combination, when differences among means occurred, means not identified by a same letter are different, based on repeated measures ANOVA of log-transformed values followed by Tukey multiple comparison test, $\alpha = 0.05$. For unburned, $n = 2, 3, 4$ experimental units and for wildfire, $n = 3, 3, 6$ experimental units for unthinned, thinned, and clear-cut, respectively.

in the order of analysis to ensure there was no bias during analysis. Subsamples were analyzed for total Hg with a Teledyne Hydra-C mercury analyzer (Teledyne Leeman Laboratories, Hudson, NH). Each subsample was analyzed in duplicate, and subsamples were rerun if the relative percent difference was greater than 10%. Results for standard reference materials averaged 95% of expected value for NIST 1547 peach leaves and 96% of expected value for NIST 2702 marine sediments. The consistency of values from the unthinned unburned composites (Figure 1) that were collected over a twenty-year period provides strong support that collection and storage methods did not influence interpretation of results. As reported in Homann et al.³⁰ composites were also analyzed for total C and N with a Thermo NC 1112 Analyzer (CE Elantech, Inc., Lakewood, New Jersey).

Soil Hg and C pools (mass m^{-2}) were calculated by the comparable layer approach.^{30,33,34} This approach accounts for mineral soil contamination of O layer samples, changes in soil volume, and soil loss resulting from fire. In this approach, soil pools are standardized by mass of soil inorganic matter per unit area rather than by depth. In this study, we defined layers based on mass of cumulative <4 mm inorganic matter: A horizon = 0–30 kg m^{-2} , B1 horizon = 30–90 kg m^{-2} , and B2 horizon = 90–180 kg m^{-2} .

Ancillary measurements of surface fine wood (1–10 cm diameter), coarse wood (>10 cm diameter), shrubs, and trees are detailed in Homann et al.^{30,34} and Bormann et al.³⁸ Tree species and diameters were measured pre and postfire on five 18-m by 18-m plots per experimental unit. Natural tree mortality, not associated with fire, occurred on unburned units. Mortality due to fire on the burned units was estimated as total mortality minus natural mortality not associated with fire.

Wildfire consumption of surface fine and coarse wood was calculated as mass at time of fire minus mass 1-year postfire.

Measurements from 2–3 years prefire and 1-year postfire were taken from Homann et al.³⁰ Over the period from prefire to postfire measurements, natural loss of fine wood occurred on unburned clear-cut units. The fine-wood fractional loss rate on these units was calculated as (prefire mass–postfire mass) divided by prefire mass and then divided by number of years between mass measurements; the resulting fine-wood fractional loss, not associated with fire, was 0.086 per year. This loss rate was used on the burned clear-cut units to estimate fine wood mass at time of wildfire, calculated as prefire mass $\times (1 - 0.086 \times \text{years between prefire measurement and wildfire})$. For fine wood in other treatments and coarse wood in all treatments, unburned treatments showed no change from prefire to postfire, and mass at time of fire was assumed to be equal to prefire. The Hg pools were calculated as mass per m^2 times the following concentrations. Composite fine wood samples from several experimental units in 2003 and 2010 were analyzed for Hg concentration with a Teledyne Hydra-C mercury analyzer. The average Hg concentration (mean 6.7 $\mu\text{g kg}^{-1}$, se = 1.0, $n = 12$ samples) was applied to all fine wood masses. The Hg concentration of coarse wood was assumed to be 2 $\mu\text{g kg}^{-1}$, based on bole data from a conifer forest site (mean 2, s.d. 1 $\mu\text{g Hg kg}^{-1}$).⁹ Prefire shrub masses on burned units were assumed to equal 2003 measurements taken on parallel unburned units.³⁴ Because field observations indicated that few shrubs survived the fire on the burned units, postfire shrubs were not measured and were assumed to be zero. We assumed an Hg concentration of 9 $\mu\text{g kg}^{-1}$ for shrubs, based on shrub data from the Little Valley, Nevada site.⁹

Statistical Analysis. To compare characteristics among treatments, single-factor ANOVA was followed by Tukey multiple comparison test at $\alpha = 0.05$. To evaluate temporal change (prefire, 1-yr postfire, 9 yr postfire) in each treatment–burn category, repeated measures ANOVA of log-transformed

Table 2. Soil Characteristics [Mean (se), for $n = 21$ Experimental Units] Prior to Treatments and Wildfire in Conifer Forest, Rogue River–Siskiyou National Forest, Oregon

horizon	thickness (mm)	rock mass (kg m ⁻²)	soil mass (kg m ⁻²)	soil bulk density (g cm ⁻³)	soil C conc. (g kg ⁻¹)	soil N conc. (g kg ⁻¹)	soil Hg conc. (μg kg ⁻¹)	soil Hg:C (μg g ⁻¹)
O ^a	nd ^b	1.1 (0.2)	2.2 (0.1)	nd	466 ^c	9.4 (0.22)	196 (9)	0.26 (0.03)
A	66 (3)	21.5 (1.5)	33.7 (0.2)	0.62 (0.03)	57 (2)	1.8 (0.07)	85 (5)	1.54 (0.05)
B1	97 (2)	34.6 (1.8)	64.5 (0.2)	0.79 (0.02)	37 (1)	1.3 (0.04)	82 (5)	2.64 (0.15)
B2	134 (3)	51.9 (3.2)	94.9 (0.1)	0.85 (0.02)	27 (1)	1.1 (0.03)	98 (7)	4.32 (0.30)

^aFor O horizon, soil values refer to the material remaining after removing >4 mm rocks and >6.4 mm diameter woody debris. For A, B1, and B2 horizons, soil values refer to the <4 mm fraction, and the horizons are defined by cumulative soil inorganic masses: A horizon, 0–30 kg m⁻²; B1 horizon, 30–90 kg m⁻²; B2 horizon, 90–180 kg m⁻². ^bnd = not determined. ^cC concentration of O horizon is defined as 466 g kg⁻¹.

data was followed by Tukey multiple comparison test at $\alpha = 0.05$. Pearson correlation was performed among selected continuous variables.

RESULTS AND DISCUSSION

Forest and Soil Characteristics. Aboveground forest structural characteristics varied among treatments (Table 1). Conifer foliar and woody biomass decreased in the order unthinned > thinned > clear-cut, while surface fine wood had the opposite order. The soil Hg concentrations decreased from O horizon to mineral soil (Table 2). The concentrations fell within the range of uncontaminated forests in the western U.S.A.⁹ The O horizon concentration was higher, while the mineral soil concentrations were lower, than those in a Douglas-fir forest in the Cascade Mountains of Washington state.⁷

Soil Hg in Absence of Recent Fire. In the absence of wildfire, O-horizon Hg mass declined by 60% in the clear-cut over the two-decade study (Figure 1). This was due to a 59% decrease in Hg concentration and nearly constant soil mass. A similar decline in O-horizon Hg concentration in a Pennsylvania oak forest was attributed to reduced Hg deposition associated with declining regional and local Hg emissions.³ This is substantiated by trends in Hg concentrations in precipitation, which decreased in the Northeast U.S.A. from 1996 to 2010.^{4,5} In the Western U.S.A., however, temporal changes in atmospheric Hg are spatially variable.⁵ To determine possible changes in our region, we examined Hg concentrations in precipitation at a National Atmospheric Deposition Program Mercury Deposition Network site³⁹ located at the H.J. Andrews Experimental Forest, 260 km to the northeast of our study site, and observed no evidence of change from 2002 through 2010. On the basis of this observation and the lack of statistically significant decrease in thinned and unthinned O-horizon Hg (Figure 1), we speculate that a regional decrease in atmospheric Hg is not a major cause of the decline in clear-cut O horizon Hg.

Alternatively, the decline in O-horizon Hg in the clear-cut may be due to reduced atmospheric deposition within only that treatment. Total deposition consists of precipitation, dry deposition of gas,⁴⁰ dry deposition of particles,¹⁵ and interception of fog and cloudwater.^{41–43} The interaction between Hg and the vegetative canopy influences the latter three processes of atmospheric deposition.^{13,15,44–48} Dry deposition of methyl mercury in open areas is only one-third the amount in forest, as indicated by throughfall and litterfall studies.^{12,13} Deposition of particulate Hg is less in open areas than in conifer forest.¹⁵ Cloudwater Hg deposition likely varies with forest characteristics.⁴³ Along upwelling areas of the Pacific coast, methyl mercury formed in the ocean may contribute to

fogwater Hg.⁴² In the Adirondack Mountains of New York, O-horizon Hg concentration was positively correlated with total Hg deposition.⁴³ At our site, the smaller-stature vegetation in the clear-cut³⁸ may be less effective in intercepting dry and cloudwater deposition than the mature forest, leading to lower total Hg deposition and a smaller O-horizon Hg pool.

A third possible mechanism for the decline of Hg in the clear-cut is enhanced Hg emissions caused by a change in the soil microclimate. Soil Hg emissions are positively related with temperature^{49–51} and UV–B radiation,^{49,50} both of which could be increased at the soil surface from clearcutting. Mazur et al.⁵² attributed increased Hg emissions following harvest and biomass removal to enhanced photoreduction in the low-shade environment. Soil Hg emissions increased immediately after cutting forest canopy.⁵³ The duration of a postharvest effect and the magnitude of total Hg emissions remain to be determined. In addition, the difference in vegetation species between mature forest and clear-cut³⁸ may create different amounts and quality of litterfall that decompose at different rates⁵⁴ and influence microbial processing of Hg.

Soil Hg Changes Associated with Wildfire. Prefire O-horizon Hg mass was ~0.4 mg m⁻² (Figure 1). Overall, wildfire decreased O-horizon Hg substantially, due to nearly total consumption of the O horizon,³⁰ but the treatments demonstrated different patterns. In unthinned, wildfire decreased O-horizon Hg mass to 12% of its prefire value, then litterfall of fire-killed needles from overstory conifers added 10% during the first year postfire, resulting in a net mass of 22% of its prefire value 1-year postfire (Figure 1). Similarly in thinned, wildfire decreased O-horizon Hg mass to 2% of its prefire value, fire-killed needles added 5%, for a net amount of 7% 1-year postfire. In contrast, the clear-cut lacked an overstory to contribute fire-killed needles and resulted in 2% of prefire value at 1-year postfire; then, in the subsequent 8 years, O-horizon Hg mass increased to 6% of prefire value as O-horizon recovery began,³⁰ concomitant with vegetation re-establishment.³⁸ Similarly, postfire soil Hg concentration increased with recovery of chaparral vegetation in California.¹⁷

Prefire A-horizon Hg mass was ~3 mg m⁻² (Figure 1). No effect of wildfire was observed in the unthinned treatment, but wildfire decreased the A-horizon Hg mass by 50% in thinned and clear-cut (Figure 1), due to 26% loss in soil mass combined with a 26% lower Hg concentration. The loss in soil mass may be due to a combination of combustion, fire-driven convective erosion, and postfire wind and water erosion.³³ The lower Hg concentration may occur from two processes. Engle et al.²³ suggested mineral soil Hg was volatilized from exposure to elevated temperatures during fire. For the upper 2 cm of mineral soil from a eucalypt forest, there was no difference in total Hg between soil samples heated to 40 or 150 °C;⁵⁵ but,

the soil lost 59% of its total Hg between 150 and 350 °C.⁵⁵ Temperatures in surface mineral soil can reach this latter range during fire^{56–58} and would likely promote substantial Hg volatilization. Alternatively, the loss of a high-Hg upper portion of the horizon would leave behind a residual portion that had lower Hg concentration. But, this would require a substantial vertical gradient in Hg concentration, which was not observed among mineral soil horizons (Table 2).

The wildfire-associated soil Hg loss from O plus A horizons was four times as great in thinned and clear-cut as in unthinned treatment (Figure 2). This demonstrates a unique influence of

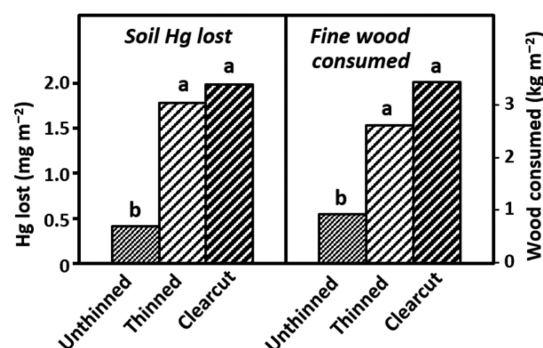


Figure 2. Mean wildfire-associated losses of soil Hg and surface fine wood consumed during wildfire for different prefire treatments (unthinned, thinned, clear-cut), Rogue River–Siskiyou National Forest, Oregon. The soil Hg losses occurred from only O horizon in unthinned, but from O plus A soil horizons in thinned and clear-cut. Within each frame, means not identified by a same letter are different, based on ANOVA of log-transformed values followed by Tukey multiple comparison test, $\alpha = 0.05$. $n = 3, 3,$ and 6 experimental units for unthinned, thinned, and clear-cut, respectively.

prefire disturbance on wildfire-associated soil Hg losses and isolates the effect of aboveground forest structure. Prefire experimental manipulations did not influence prefire O-horizon Hg pools and had minor influence on the A-horizon Hg pool only in the clear-cut, as confirmed by comparing pretreatment and 6-yr posttreatment values for the unburned units (Figure 1). Instead, treatments influenced aboveground forest structure (Table 1) that subsequently affected how wildfire impacted soil Hg losses, with greater losses occurring in clear-cut and thinned manipulations. In contrast, in Jack pine forest, prefire blowdown and postblowdown disturbance altered both forest structure and forest floor Hg pools, which jointly resulted in greater wildfire-associated losses.²⁷ Both mechanisms—alteration of prefire forest structure and alteration of prefire soil Hg pool size—are important to recognize and consider when evaluating wildfire influences.

The soil Hg losses fall within the range of other wildfire studies and extend above the range of prescribed fire studies (Table 3). The variation in Hg loss among the different studies may be due to the magnitudes of prefire soil Hg pools and characteristics of the fires. In contrast with other studies, this study differentiates between Hg losses from O horizon and from mineral soil. In the clear-cut and thinned forest, ~80% of soil Hg that was lost was from mineral soil and only ~20% from O horizon (Figure 1). Other studies either have not differentiated between losses from O horizon and mineral soil, have evaluated only the O horizon, have assessed only the mineral soil, or have not found losses from mineral soil (Table 3). For example, in a pine–deciduous forest, mineral-soil Hg pools did not differ among various burn severity classes, including unburned sites.⁵⁹

This study measured soil Hg twice after fire and found only small recovery of soil Hg pools during the decade following fire (Figure 1). This suggests that measuring Hg pools several years

Table 3. Fire-Associated Soil Hg Loss from Forest Organic (O) and Mineral (M) Horizons

location	sampling approach	years since fire	forest type	soil Hg loss (mg m^{-2})	horizon	reference
Southwest Oregon	pre- and postfire	0	wildfire			
			Douglas-fir, mature	0.5	O	current study
				0	M	
			Douglas-fir, thinned	0.4	O	
				1.4	M	
Douglas-fir, clear-cut	0.4	O				
	1.6	M				
Sierra Nevada Mtn Jackson, Wyoming	pre- and postfire	1	pine and fir	0.26	O	23
			conifer	0.74 to 2.5	O + M	25
				2–3	0.36 to 1.3	O + M
				2–3	0.41	O + M
Eastern Washington	postfire only	3	pine and fir	0.67	O + M	26
			meadow	0.11	O + M	
Northern Minnesota	postfire only	1	jack pine, control	0	O	27
			jack pine, blowdown	0.9	O	
Northern Minnesota	postfire only	0–1	pine and deciduous	0.3	O	59
Czech Republic	postfire only	<2	spruce and pine	7.5	O	24
Portugal	postfire only	0	eucalypt	0.1	M	55
			prescribed fire			
Sierra Nevada Mtn	postfire only	1	pine and fir	0.36	O	23
Sierra Nevada Mtn	postfire only	5	pine and fir	0.1	O	60
Central Alaska	pre- and postfire	0	black spruce	0.2	O	61
Florida	postfire only	0.5	longleaf pine	0.8	O	62
Amazon, Brazil	pre- and postfire	0	rain forest	0.3	O	63

after fire, as performed in several studies (Table 3), introduces little error in estimates of fire-induced soil Hg loss, provided postfire erosion losses are small. However, measurement methods may introduce uncertainty. This study determined soil change by comparing prefire and postfire samples, an approach used in few other studies (Table 3). In contrast, most estimates of fire-associated soil Hg loss are based on postfire-only sampling,^{24–27} which requires the assumption that an unburned site represents prefire conditions. Although this assumption has been scrutinized and inappropriate comparisons between burned and unburned sites are sometimes identified,²⁶ it introduces unknown uncertainty into the estimates of soil Hg losses. Conversely, this assumption and its associated uncertainty are nonexistent when prefire and postfire soil samples from the same location are compared, as was performed in this study.

The soil Hg losses were nearly 2 orders of magnitude greater than Hg losses from combustion of surface wood and understory vegetation. The Hg losses from surface fine wood combustion were estimated to be 0.006, 0.017, and 0.023 mg m⁻², for unthinned, thinned, and clear-cut, respectively; Hg losses from surface coarse wood combustion, 0.002, 0.002, and 0.004 mg m⁻²; and Hg losses from understory combustion, 0.003, 0.003, and 0.014 mg m⁻². The Hg loss from combustion of overstory bark and wood is expected to have been very small, too, because of the ground-fire nature of the event.³² This importance of soil Hg losses, compared with losses of other ecosystem components, is consistent with observations of Sierra-Nevada pine–fir forest.²³

Across all wildfire units, soil Hg loss was positively curvilinearly related to soil C loss (Figure 3); and soil C loss,

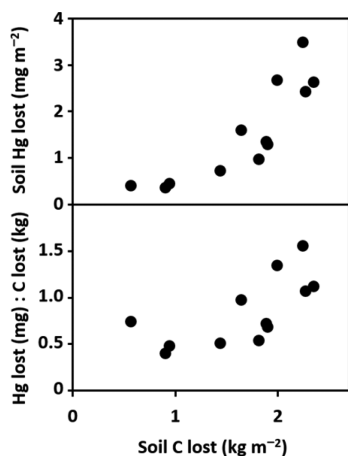


Figure 3. Relation between wildfire-associated losses of soil Hg and C across 12 experimental units that burned in the 2002 Biscuit wildfire, Rogue River–Siskiyou National Forest, Oregon. Losses were from O plus A horizons. For log(soil Hg lost) versus soil C lost, $r = 0.94$, $P < 0.0001$. For log(Hg lost/C lost) versus soil C lost, $r = 0.68$, $P = 0.014$.

an indicator of soil organic matter combustion, was positively related to the proportion of the area that reached aboveground temperatures >700 °C.³³ Higher soil C loss signifies burning deeper into the soil profile, where the Hg:C ratio is greater (Table 2). Consequently, the Hg-loss/C-loss ratio is positively related to soil C loss (Figure 3). These relations might be used in making first approximations of Hg losses where wildfire-associated soil C losses have been estimated.⁶⁴

Soil Hg Loss and Fire Characteristics. The wildfire-associated loss of soil Hg was related to several fire characteristics. The greater loss of soil Hg in thinned and clear-cut paralleled the pattern for surface fine wood that was consumed during the wildfire (Figure 2). Across all 12 wildfire units, the log-transformed soil Hg loss was positively related to surface fine wood consumed during fire ($r = 0.63$, $P = 0.03$) and to surface fine wood present at time of wildfire ($r = 0.60$, $P = 0.04$). In contrast, the log-transformed soil Hg loss was not related to surface coarse wood consumed ($r = 0.29$, $P = 0.36$) or surface coarse wood present ($r = 0.28$, $P = 0.37$).

Wildfire-associated soil Hg loss was also positively related to burn temperature. In a subset of 9 wildfire units, we have estimates of the % of area that burned at >700 °C, as indicated by extent of melting of aluminum tags.³³ The log of soil Hg loss was positively correlated with % of area burned at >700 °C ($r = 0.80$, $P < 0.01$), which was itself positively correlated with the log of surface fine wood consumed ($r = 0.88$, $P < 0.01$). The positive correlations among soil Hg loss, burn temperature, and soil organic matter combustion, as indicated by soil C loss (Figure 3), support the concept that higher soil temperatures contributed to greater Hg loss in this study. In laboratory experiments, this relation is well documented. For example, for eucalypt forest mineral soil, 59% of its total Hg was lost between 150 and 350 °C and an additional 38% between 350 and 550 °C.⁵⁵ Further, in the thermal desorption technique to characterize chemical forms of soil Hg, Hg loss increases as temperature is raised to several hundred °C.^{65,66}

The wildfire-associated soil Hg loss was related to the % conifer mortality. This variable could be assessed only in the unthinned and thinned treatments because clear-cut lacked mature trees. The relation was positive and monotonic, but not linear (Figure 4). At $>80\%$ mortality, a threshold was passed, and there was a dramatic increase in soil Hg loss.

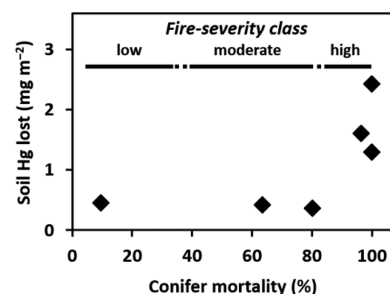


Figure 4. Relation between wildfire-associated losses of soil Hg and wildfire-associated conifer mortality across 6 unthinned and thinned experimental units that burned in the 2002 Biscuit wildfire, Rogue River–Siskiyou National Forest, Oregon. Hg losses were from O plus A horizons. Fire-severity classes are based on burn area emergency rehabilitation (BAER) descriptions.⁶⁷

Tree mortality is a key descriptor of fire-severity classes of the burn area emergency rehabilitation (BAER) assessment of the Biscuit Wildfire, with moderate severity described as 40 to 80% of trees killed and high severity as near 100% tree death.⁶⁷ When our units are classified based on conifer mortality, the ratio of soil Hg lost was 4.3:1.0 for high/moderate-low severity (Figure 4). Similarly, in a Wyoming forest wildfire, soil Hg lost varied among fire severity classes.²⁵ In that study, fire severity classes were based on both aboveground indicators (tree trunk burn height, and the proportion of burned canopy) and soil assessment (the depth of ash on the soil surface). The ratio of

soil Hg lost was 3.4:2.8:1.0 for high/moderate/low severity in conifer forest, and 3.8:1.0 for moderate/low severity in aspen forest. In contrast, in a Washington state wildfire study that used the same fire-severity indicators, there was no difference in soil Hg lost among high, moderate, and low severities in conifer forest.²⁶ Further, in studies of two Minnesota wildfires, fire severity was designated based primarily on postfire forest floor and mineral soil characteristics; no difference in postfire soil Hg pool was observed among the burned fire-severity classes.^{27,59} When fire severity classes were based on canopy color, the severely burned class had lower forest floor Hg pool than lightly burned and very severely burned classes.⁵⁹

Our study indicates aboveground forest structure influenced the amount of soil Hg that was lost due to wildfire. In particular, greater amounts of prefire surface fine wood resulted in greater fine wood consumption, leading to higher aboveground fire severity and greater loss of soil Hg. Of the 200 000 ha area within the Biscuit wildfire perimeter, 16% was classified as high severity, 23% as moderate, and 42% as low, based on the BAER fire severity map.⁶⁸ Applying our average soil Hg losses for these classes yields 1110 kg Hg released by the Biscuit wildfire. Although some loss may be due to postfire erosion, this total loss is more than four times the annual atmospheric Hg emissions from human activities in Oregon,⁶⁹ the location of the Biscuit wildfire, and substantiates the importance of wildfires to Hg emissions in the U.S.A.⁷⁰ The chemical form of Hg emitted was not determined in this study, but atmospheric analysis following Oregon and California wildfires indicates elemental Hg is dominant and particulate Hg makes up 15% of the Hg that is released.¹⁶ In future decades, forest wildfire extent is projected to rise dramatically in the western U.S.A. and Canada, with 24 to 169% increases in burned area forecast for various regions.^{19–21} Fire severity may also increase,⁷¹ and forest management techniques may be implemented to affect wildfire behavior.⁷² Integration of these projections with our assessment of wildfire-associated Hg losses will improve the understanding of the importance of wildfire in regional and global Hg cycles.

AUTHOR INFORMATION

Corresponding Author

*Phone: 360-650-7585; fax: 360-650-7284; e-mail: Peter.Homann@wwu.edu (P.S.H.).

Notes

The authors declare no competing financial interest.

ACKNOWLEDGMENTS

This paper is a contribution of the USDA Forest Service, Pacific Northwest Research Station's Long-Term Ecosystem Productivity Program. Support for pre- and posttreatment, and postwildfire sampling and analysis came from the Research Station, the US Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon (Interagency Agreement DW 12936179), the Joint Fire Sciences Program (Grants 03-2-3-09 and 10-1-10-18), Western Washington University, the National Commission for Science on Sustainable Forestry (Grant C4), and the Rogue River—Siskiyou National Forest. We acknowledge the hard work of many individuals, including former LTEP experiment leaders Susan Little and Mike Amaranthus; agreement leads from Oregon State University, Kermit Cromack Jr. and Mark Harmon. This work would not have happened without the professional field

and laboratory assistants from past years: Tom Bell, Aurore Chauvry, Matt Cowall, Colin Edgar, Laura Fabrey Stevens, Nate France, Nick Leahy, Kristina Muscutt, Suzanne Remillard, Vanessa Spini, Chris Stevens, Kyle Swanson, and Dave Woodruff; and from recent years: Amy Barnhart, Dylan Burgess, Nick Daniel, Martyn Davies, Emma Garner, Tim Martin, Kylie Meyer, and Alex Van Loo. We thank Julia Denning for preliminary study of soil mercury and Prof. Paul Gremillion for conducting mercury analyses at the Environmental Mercury Laboratory, Northern Arizona University.

REFERENCES

- (1) Selin, N. E. Global biogeochemical cycling of mercury: A review. *Annu. Rev. Environ. Resour.* **2009**, *34*, 43–63.
- (2) Grigal, D. F. Mercury sequestration in forests and peatlands: A review. *J. Environ. Qual.* **2003**, *32*, 393–405.
- (3) McClenahan, J. R.; Hutnik, R. J.; Davis, D. D. Spatial and temporal patterns of bioindicator mercury in Pennsylvania oak forest. *J. Environ. Qual.* **2013**, *42*, 305–311.
- (4) Prestbo, E. M.; Gay, D. A. Wet deposition of mercury in the US and Canada, 1996–2005: Results and analysis of the NADP mercury deposition network (MDN). *Atmos. Environ.* **2009**, *43*, 4223–4233.
- (5) Zhang, Y. X.; Jaegle, L. Decreases in mercury wet deposition over the United States during 2004–2010: Roles of domestic and global background emission reductions. *Atmosphere* **2013**, *4*, 113–131.
- (6) Cole, A. S.; Steffen, A.; Eckley, C. S.; Narayan, J.; Pilote, M.; Tordon, R.; Graydon, J. A.; St Louis, V. L.; Xu, X. H.; Branfireun, B. A. A survey of mercury in air and precipitation across Canada: Patterns and trends. *Atmosphere* **2014**, *5*, 635–668.
- (7) Gurnesa, G. A.; Schmidt, I. K.; Gundersen, P.; Vesterdal, L. Soil carbon accumulation and nitrogen retention traits of four tree species grown in common gardens. *For. Ecol. Manage.* **2013**, *309*, 47–57.
- (8) Barcena, T. G.; Gundersen, P.; Vesterdal, L. Afforestation effects on SOC in former cropland: oak and spruce chronosequences resampled after 13 years. *Global Change Biol.* **2014**, *20*, 2938–2952.
- (9) Obrist, D.; Johnson, D. W.; Lindberg, S. E.; Luo, Y.; Hararuk, O.; Bracho, R.; Battles, J. J.; Dail, D. B.; Edmonds, R. L.; Monson, R. K.; Ollinger, S. V.; Pallardy, S. G.; Pregitzer, K. S.; Todd, D. E. Mercury distribution across 14 U.S. forests. Part I: Spatial patterns of concentrations in biomass, litter, and soils. *Environ. Sci. Technol.* **2011**, *45*, 3974–3981.
- (10) Yu, X.; Driscoll, C. T.; Warby, R. A. F.; Montesdeoca, M.; Johnson, C. E. Soil mercury and its response to atmospheric mercury deposition across the northeastern United States. *Ecol. Appl.* **2014**, *24*, 812–822.
- (11) Obrist, D.; Johnson, D. W.; Edmonds, R. L. Effects of vegetation type on mercury concentrations and pools in two adjacent coniferous and deciduous forests. *J. Plant Nutr. Soil Sci.* **2012**, *175*, 68–77.
- (12) Munthe, J.; Hultberg, H.; Iverfeldt, A. Mechanisms of deposition of methylmercury and mercury to coniferous forests. *Water, Air, Soil Pollut.* **1995**, *80*, 363–371.
- (13) Mowat, L. D.; St Louis, V. L.; Graydon, J. A.; Lehnher, I. Influence of forest canopies on the deposition of methylmercury to boreal ecosystem watersheds. *Environ. Sci. Technol.* **2011**, *45*, 5178–5185.
- (14) Risch, M. R.; DeWild, J. F.; Krabbenhoft, D. P.; Kolka, R. K.; Zhang, L. M. Litterfall mercury dry deposition in the eastern USA. *Environ. Pollut.* **2012**, *161*, 284–290.
- (15) Witt, E. L.; Kolka, R. K.; Nater, E. A.; Wickman, T. R. Forest fire effects on mercury deposition in the boreal forest. *Environ. Sci. Technol.* **2009**, *43*, 1776–1782.
- (16) Finley, B. D.; Swartzendruber, P. C.; Jaffe, D. A. Particulate mercury emissions in regional wildfire plumes observed at the Mount Bachelor Observatory. *Atmos. Environ.* **2009**, *43*, 6074–6083.
- (17) Burke, M. P.; Hogue, T. S.; Ferreira, M.; Mendez, C. B.; Navarro, B.; Lopez, S.; Jay, J. A. The effect of wildfire on soil mercury concentrations in southern California watersheds. *Water, Air, Soil Pollut.* **2010**, *212*, 369–385.

- (18) Kelly, E. N.; Schindler, D.; St Louis, V. L.; Donald, D. B.; Vladicka, K. E. Forest fire increases mercury accumulation by fishes via food web restructuring and increased mercury inputs. *Proc. Natl. Acad. Sci. U. S. A.* **2006**, *103*, 19380–19385.
- (19) Spracklen, D. V.; Mickley, L. J.; Logan, J. A.; Hudman, R. C.; Yevich, R.; Flannigan, M. D.; Westerling, A. L. Impacts of climate change from 2000 to 2050 on wildfire activity and carbonaceous aerosol concentrations in the western United States. *J. Geophys. Res.* **2009**, *114*, D20301.
- (20) Nitschke, C. R.; Innes, J. L. Potential effect of climate change on observed fire regimes in the Cordilleran forests of south-central interior, British Columbia. *Clim. Change* **2013**, *116*, 579–591.
- (21) Yue, X.; Mickley, L. J.; Logan, J. A.; Kaplan, J. O. Ensemble projections of wildfire activity and carbonaceous aerosol concentrations over the western United States in the mid-21st century. *Atmos. Environ.* **2013**, *77*, 767–780.
- (22) Turetsky, M. R.; Harden, J. W.; Friedli, H. R.; Flannigan, M.; Payne, N.; Crock, J.; Radke, L. Wildfires threaten mercury stocks in northern soils. *Geophys. Res. Lett.* **2006**, *33*, L16403.
- (23) Engle, M. A.; Gustin, M. S.; Johnson, D. W.; Murphy, J. F.; Miller, W. W.; Walker, R. F.; Wright, J.; Markee, M. Mercury distribution in two Sierran forest and one desert sagebrush steppe ecosystems and the effects of fire. *Sci. Total Environ.* **2006**, *367*, 222–233.
- (24) Navrátil, T.; Hojdová, M.; Rohovec, J.; Penížek, V.; Vařilová, Z. Effect of fire on pools of mercury in forest soil, central Europe. *Bull. Environ. Contam. Toxicol.* **2009**, *83*, 269–274.
- (25) Biswas, A.; Blum, J. D.; Klaue, B.; Keeler, G. J. The release of mercury from Rocky Mountain forest fires. *Global Biogeochem. Cycles* **2007**, *21*, GB1002.
- (26) Biswas, A.; Blum, J. D.; Keeler, G. J. Mercury storage in surface soils in a central Washington forest and estimated release during the 2001 Rex Creek Fire. *Sci. Total Environ.* **2008**, *404*, 129–138.
- (27) Mitchell, C. P. J.; Kolka, R. K.; Fraver, S. Singular and combined effects of blowdown, salvage logging, and wildfire on forest floor and soil mercury pools. *Environ. Sci. Technol.* **2012**, *46*, 7963–7970.
- (28) Woodruff, L. G.; Cannon, W. F. Immediate and long-term fire effects on total mercury in forests soils of Northeastern Minnesota. *Environ. Sci. Technol.* **2010**, *44*, 5371–5376.
- (29) McElhinny, C.; Gibbons, P.; Brack, C.; Bauhus, J. Forest and woodland stand structural complexity: Its definition and measurement. *For. Ecol. Manage.* **2005**, *218*, 1–24.
- (30) Homann, P. S.; Bormann, B. T.; Morrisette, B. A.; Darbyshire, R. L. Postwildfire soil trajectory linked to prefire ecosystem structure in Douglas-fir forest. *Ecosystems* **2015**, *18*, 260–273.
- (31) Little, R. L.; Peterson, D. L.; Silsbee, D. G.; Shainsky, L. J.; Bednar, L. F. Radial growth patterns and the effects of climate on second-growth Douglas-fir (*Pseudotsuga menziesii*) in the Siskiyou Mountains, Oregon. *Can. J. For. Res.* **1995**, *25*, 724–35.
- (32) Raymond, C. L.; Peterson, D. L. Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. *Can. J. For. Res.* **2005**, *35*, 2981–95.
- (33) Bormann, B. T.; Homann, P. S.; Darbyshire, R. L.; Morrisette, B. A. Intense forest wildfire sharply reduces mineral soil C and N: The first direct evidence. *Can. J. For. Res.* **2008**, *38*, 2771–2783.
- (34) Homann, P. S.; Bormann, B. T.; Darbyshire, R. L.; Morrisette, B. A. Forest soil carbon and nitrogen losses associated with wildfire and prescribed fire. *Soil Sci. Soc. Am. J.* **2011**, *75*, 1926–34.
- (35) Homann, P. S.; Bormann, B. T.; Boyle, J. R. Detecting treatment differences in soil carbon and nitrogen resulting from forest manipulations. *Soil Sci. Soc. Am. J.* **2001**, *65*, 463–469.
- (36) Homann, P. S.; Bormann, B. T.; Boyle, J. R.; Darbyshire, R. L.; Bigley, R. Soil C and N minimum detectable changes and treatment differences in a multi-treatment forest experiment. *For. Ecol. Manage.* **2008**, *255*, 1724–1734.
- (37) Hojdová, M.; Rohovec, J.; Chrástný, V.; Penížek, V. Navrátil. The influence of sample drying procedures on mercury concentrations analyzed in soils. *Bull. Environ. Contam. Toxicol.* **2015**, *94*, 570–576.
- (38) Bormann, B. T.; Darbyshire, R. L.; Homann, P. S.; Morrisette, B. A.; Little, S. N. Managing early succession for biodiversity and long-term productivity of conifer forests in southwestern Oregon. *For. Ecol. Manage.* **2015**, *340*, 114–125.
- (39) NADP. National Atmospheric Deposition Network (NADP)-Mercury Deposition Network. <http://nadp.sws.uiuc.edu/data/MDN/> (accessed March 2015).
- (40) Laacouri, A.; Nater, E. A.; Kolka, R. K. Distribution and uptake dynamics of mercury in leaves of common deciduous tree species in Minnesota, USA. *Environ. Sci. Technol.* **2013**, *47*, 10462–10470.
- (41) Lawson, S. T.; Scherbatskoy, T. D.; Malcolm, E. G.; Keeler, G. J. Cloud water and throughfall deposition of mercury and trace elements in a high elevation spruce-fir forest at Mt. Mansfield, Vermont. *J. Environ. Monit.* **2003**, *5*, 578–583.
- (42) Weiss-Penzias, P. S.; Ortiz, C.; Acosta, R. P.; Heim, W.; Ryan, J. P.; Fernandez, D.; Collett, J. L.; Flegal, A. R. Total and monomethyl mercury in fog water from the central California coast. *Geophys. Res. Lett.* **2012**, *39*, L03804.
- (43) Blackwell, B. D.; Driscoll, C. T. Deposition of mercury in forests along a montane elevation gradient. *Environ. Sci. Technol.* **2015**, *49*, 5363–5370.
- (44) Rea, A. W.; Keeler, G. J.; Scherbatskoy, T. The deposition of mercury in throughfall and litterfall in the Lake Champlain watershed: A short-term study. *Atmos. Environ.* **1996**, *30*, 3257–3263.
- (45) St Louis, V. L.; Rudd, J. W. M.; Kelly, C. A.; Hall, B. D.; Rolffus, K. R.; Scott, K. J.; Lindberg, S. E.; Dong, W. Importance of the forest canopy to fluxes of methyl mercury and total mercury to boreal ecosystems. *Environ. Sci. Technol.* **2001**, *35*, 3089–3098.
- (46) Zhang, L. M.; Wright, L. P.; Blanchard, P. A review of current knowledge concerning dry deposition of atmospheric mercury. *Atmos. Environ.* **2009**, *43*, 5853–5864.
- (47) Yu, X.; Driscoll, C. T.; Huang, J. Y.; Holsen, T. M.; Blackwell, B. D. Modeling and mapping of atmospheric mercury deposition in Adirondack Park, New York. *PLoS One* **2013**, *8*, e59322.
- (48) Blackwell, B. D.; Driscoll, C. T.; Maxwell, J. A.; Holsen, T. M. Changing climate alters inputs and pathways of mercury deposition to forested ecosystems. *Biogeochemistry* **2014**, *119*, 215–228.
- (49) Choi, H. D.; Holsen, T. M. Gaseous mercury emissions from unsterilized and sterilized soils: The effect of temperature and UV radiation. *Environ. Pollut.* **2009**, *157*, 1673–1678.
- (50) Choi, H. D.; Holsen, T. M. Gaseous mercury fluxes from the forest floor of the Adirondacks. *Environ. Pollut.* **2009**, *157*, 592–600.
- (51) Kim, K. H.; Yoon, H. O.; Jung, M. C.; Oh, J. M.; Brown, R. J. C. A simple approach for measuring emission patterns of vapor phase mercury under temperature-controlled conditions from soil. *Sci. World J.* **2012**, Article 940413; DOI 10.1100/2012/940413.
- (52) Mazur, M.; Mitchell, C. P. J.; Eckley, C. S.; Eggert, S. L.; Kolka, R. K.; Sebestyen, S. D.; Swain, E. B. Gaseous mercury fluxes from forest soils in response to forest harvesting intensity: A field manipulation experiment. *Sci. Total Environ.* **2014**, *496*, 678–687.
- (53) Carpi, A.; Fostier, A. H.; Orta, O. R.; dos Santos, J. C.; Gittings, M. Gaseous mercury emissions from soil following forest loss and land use changes: Field experiments in the United States and Brazil. *Atmos. Environ.* **2014**, *96*, 423–429.
- (54) Valachovic, Y. S.; Caldwell, B. A.; Cromack, K., Jr.; Griffiths, R. P. Leaf litter chemistry controls on decomposition of Pacific Northwest trees and woody shrubs. *Can. J. For. Res.* **2004**, *34*, 2131–2147.
- (55) Campos, I.; Vale, C.; Abrantes, N.; Keizer, J. J.; Pereira, P. Effects of wildfire on mercury mobilization in eucalypt and pine forests. *Catena* **2015**, *131*, 149–159.
- (56) Stooft, C. R.; Moore, D.; Fernandes, P. M.; Stoorvogel, J. J.; Fernandes, R. E. S.; Ferreira, A. J. D.; Ritsema, C. J. Hot fire, cool soil. *Geophys. Res. Lett.* **2013**, *40*, 1534–1539.
- (57) Certini, G. Effects of fire on properties of forest soils: a review. *Oecologia* **2005**, *143*, 1–10.
- (58) Granged, A. J. P.; Jordan, A.; Zavala, L. M.; Munoz-Rojas, M.; Mataix-Solera, J. Short-term effects of experimental fire for a soil under eucalyptus forest (SE Australia). *Geoderma* **2011**, *167–168*, 125–134.

(59) Kolka, R.; Sturtevant, B.; Townsend, P.; Miesel, J.; Wolter, P.; Fraver, S.; DeSutter, T. Post-fire comparisons of forest floor and soil carbon, nitrogen, and mercury pools with fire severity indices. *Soil Sci. Soc. Am. J.* **2014**, *78*, S58–S65.

(60) Obrist, D.; Johnson, D. W.; Lindberg, S. E. Mercury concentrations and pools in four Sierra Nevada forest sites, and relationships to organic carbon and nitrogen. *Biogeosciences* **2009**, *6*, 765–777.

(61) Harden, J. W.; Neff, J. C.; Sandberg, D. V.; Turetsky, M. R.; Ottmar, R.; Gleixner, G.; Fries, T. L.; Manies, K. L. Chemistry of burning the forest floor during the FROSTFIRE experimental burn, interior Alaska, 1999. *Global Biogeochem. Cycles* **2004**, *18*, GB3014.

(62) DiCosto, R. J.; Callahan, M. A., Jr.; Stanturf, J. A. Atmospheric deposition and re-emission of mercury estimated in a prescribed forest-fire experiment in Florida, USA. *Water, Air, Soil Pollut.* **2006**, *176*, 77–91.

(63) Melendez-Perez, J. J.; Fostier, A. H.; Carvalho, J. A.; Windmoller, C. C.; Santos, J. C.; Carpi, A. Soil and biomass mercury emissions during a prescribed fire in the Amazonian rain forest. *Atmos. Environ.* **2014**, *96*, 415–422.

(64) Baird, M.; Zabowski, D.; Everett, R. L. Wildfire effects on carbon and nitrogen in inland coniferous forests. *Plant Soil* **1999**, *209*, 233–243.

(65) Biester, H.; Scholz, C. Determination of mercury binding forms in contaminated soils: Mercury pyrolysis versus sequential extractions. *Environ. Sci. Technol.* **1997**, *31*, 233–239.

(66) Reis, A. T.; Coelho, J. P.; Rucandio, I.; Davidson, C. M.; Duarte, A. C.; Pereira, E. Thermo-desorption: A valid tool for mercury speciation in soils and sediments? *Geoderma* **2015**, *237*, 98–104.

(67) Azuma, D. L.; Donnegan, J.; Gedney, D. Southwest Oregon Biscuit fire: An analysis of forest resources and fire severity. Pacific Northwest Research Station Res. Paper PNW-RP-560. USDA Forest Service: Portland, OR, 2004.

(68) Campbell, J.; Donato, D.; Azuma, D.; Law, B. Pyrogenic carbon emission from a large wildfire in Oregon, United States. *J. Geophys. Res.* **2007**, *112*, G04014.

(69) US EPA. The 2011 National Emissions Inventory; <http://www.epa.gov/ttn/chief/net/2011inventory.html>. (accessed March 2015).

(70) Wiedinmyer, C.; Friedli, H. Mercury emission estimates from fires: An initial inventory for the United States. *Environ. Sci. Technol.* **2007**, *41*, 8092–8098.

(71) Cansler, C. A.; McKenzie, D. Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade range, USA. *Ecol. Appl.* **2014**, *24*, 1037–1056.

(72) Wimberly, M. C.; Liu, Z. H. Interactions of climate, fire, and management in future forests of the Pacific Northwest. *For. Ecol. Manage.* **2014**, *327*, 270–279.