Wildfire burn severity and stream chemistry influence aquatic invertebrate and riparian avian mercury exposure in forested ecosystems

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Abstract

Terrestrial soils in forested landscapes represent some of the largest mercury (Hg) reserves globally. Wildfire can alter the storage and distribution of terrestrial-bound Hg via reemission to the atmosphere or mobilization in watersheds where it may become available for methylation and uptake into food webs. Using data associated with the 2007 Moonlight and Antelope Fires in California, we examined the long-term direct effects of wildfire burn severity on the distribution and magnitude of Hg concentrations in riparian food webs. Additionally, we quantified the cross-ecosystem transfer of Hg from aquatic invertebrate to riparian bird communities; and assessed the influence of biogeochemical, landscape variables, and ecological factors on Hg concentrations in aquatic and terrestrial food webs. Benthic macroinvertebrate methylmercury (MeHg) and riparian bird blood total mercury (THg) concentrations varied by 710- and 760-fold, respectively, and Hg concentrations were highest in predators. We found inconsistent relationships between Hg concentrations across and within taxa and guilds in response to stream chemical parameters and burn severity. Macroinvertebrate scraper MeHg concentrations were influenced by dissolved organic carbon (DOC); however, that relationship was moderated by burn severity (as burn severity increased the effect of DOC declined). Omnivorous bird Hg concentrations declined with increasing burn severity. Overall, taxa more linked to in situ energetic pathways may be more responsive to the biogeochemical processes that influence MeHg cycling. Remarkably, 8 years post-fire, we still observed evidence of burn severity influencing Hg concentrations within riparian food webs, illustrating its overarching role in altering the storage and redistribution of Hg and influencing biogeochemical processes.

Keywords Bioaccumulation · Environmental exposure · Forest · Hg · Landscape

Introduction

The long-term and ongoing emissions and atmospheric transport of mercury (Hg) have resulted in the global

deposition of Hg to aquatic and terrestrial environments (Eagles-Smith et al. 2018, Hsu-Kim et al. 2018, Orbrist et al. 2018). Once deposited, Hg can undergo a range of processes, including methylation and trophic transfer through food webs, reemission to the atmosphere, or sequestration into longer-term storage reservoirs like glaciers and sediments. Soils represent the largest global terrestrial reservoir of Hg (Grigal 2003, Obrist et al. 2012), particularly soils in heavily vegetated environments with high organic content, like conifer forests (Obrist et al. 2012). Although soils can represent a long-term reservoir of Hg deposition, the increasing frequency, intensity, and extent of wildfires (Orbrist et al. 2018) over the past decade threatens to alter the global storage and distribution of Hg through either reemission to the atmosphere or mobilization in watersheds where it may become available for methylation and uptake into food webs. Additionally, wildfires can dramatically alter

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the physical, geochemical, and ecological characteristics of watersheds (Rogers et al. 2020), potentially influencing the post-fire movement of Hg through food webs.

Combustion of soil and vegetation can result in the near complete export of Hg to the atmosphere (Friedli et al. 2003, Mailman and Bodaly 2005), although some Hg remains in the residual ash (Friedli et al. 2003). Burn severity (the degree to which a site has been altered or disrupted by fire; loosely, a product of fire intensity and residence time; NWCG National Wildfire Coordinating Group 2005) influences the extent to which Hg is released, with higher severity fires releasing more Hg from the local soil, leaving less Hg on the fire-influenced landscape (Biswas et al. 2007). However, the relative distribution of post-fire Hg in ash or soil on the burned landscape and its influence on downslope aquatic and riparian systems and associated terrestrial taxa are poorly understood. Soil carbon and dissolved organic carbon (DOC) play critical roles in the retention and transport of soil-bound Hg and its conversion to methylmercury (MeHg) when transported to aquatic systems. Dissolved organic carbon facilitates both the transport of Hg into aquatic ecosystems, as well as microbial MeHg production (Lambertsson and Nilsson 2006). Additionally, DOC influences the bioavailability of Hg and its uptake into food webs (Wiener et al.; 2006, Dutton and Fisher 2012). As such, wildfire-induced combustion of soil carbon may result in reduced DOC concentrations in downstream aquatic systems (Wei et al. 2021), which could influence Hg cycling and transport of Hg from terrestrial to aquatic systems.

Other water chemistry parameters also play important roles in net MeHg production within aquatic ecosystems. Complexing ligands such as sulfide can inhibit MeHg formation when bound to Hg^{2+} (Barrow and Cox 1992, Craig and Moreton 1985, Kim et al., (2003), Chen et al. 2020), whereas sulfate is an important terminal electron acceptor for Hg-methylating microbes (Kim et al., 2003, Mehrotra and Sedlak, 2005, Windham-Myers et al., 2009), and low pH often promotes inorganic Hg bioavailability and MeHg production. Despite the well-documented linkages between wildfire and stream biogeochemistry, there is a paucity of data on the compounding effects of wildfire on MeHg concentrations within both aquatic fauna (invertebrates) and adjacent riparian taxa (e.g., birds). Similarly, there is a paucity of studies that have examined how wildfires may have long-term effects on water chemistry and associated Hg exposure in aquatic and aquatic-dependent species (but see Banks et al. 2005, Patel et al. 2019).

In this study, we evaluated the long-term influence of burn severity associated with a large wildfire complex (Moonlight and Antelope Fires) that occurred during 2007 in the Plumas National Forest in northern California nearly eight years before our study (Fig. 1). Our goals were to (1) assess the spatial distribution and magnitude of Hg concentrations in riparian food webs within the Moonlight/Antelope fire area; (2) quantify cross-ecosystem transfer of Hg from aquatic prey into riparian bird communities; and (3) assess how the influence of chemical and landscape variables and ecological factors influenced Hg concentrations in aquatic and terrestrial food webs.

Materials and methods

Study area

The Plumas National Forest is in the northern terminus of the Sierra Nevada Mountain range in northern California (Fig. 1), spanning 4682 km^2 across Plumas, Butte, Sierra, Lassen, and Yuba counties. The Plumas National Forest is situated in the Lower Sacramento River sub-region and the North Fork Feather, East Branch North Fork Feather, Middle Fork Feather, and Upper Yuba sub-basins. It encompasses conifer forests, broad mid-elevation meadows and valleys, and steep river canyons. Streams within the forest are primarily first- through third-order (i.e., headwater) streams based on Strahler stream order. The Moonlight and Antelope fires were two adjacent contemporaneous fires with unique ignition sources that burned approximately 360 km^2 primarily within the Plumas National Forest between July and September of 2007. Wildfire intensity varied across the burn zone, with burn severity values ranging from unchanged to high based on the Monitoring Trends in Burn Severity (MTBS; U.S. Geological Survey, National Center for Earth Resources and Observation and Science and the USDA Forest Service, Geospatial Technology and Applications Center Eidenshink et al. 2007). During the time between the Moonlight and Antelope wildfires, there were no additional wildfires within the watersheds we studied (Cal Fire 2023).

Field sampling

In 2015, we selected 24 sites for both invertebrate and bird sampling throughout the range of the fire boundary and 5 sites outside the fire boundary where no fire activity occurred (Fig. 1). Site locations were chosen using a generalized random tessellation stratified sampling design, which reduces clumping compared to a simple random sample (Brown et al. 2015). Sites represented a range of fire severities (see *Landscape Variables – Burn Severity* below), from unburned/unchanged to high, and were located on first and second-order perennial streams. Sites were sampled for water chemistry and aquatic invertebrates (density, species composition, Hg concentrations) between June 1 and August 25, 2015. During 2016, we sampled an additional 29 sites for birds only. Although our study occurred over 7 years

Fig. 1 Locations of aquatic invertebrate and riparian bird mercury sampling throughout and adjacent to the 2007 Moonlight and Antelope fire boundaries within the Plumas National Forest, California during 2015–2016. The color scheme indicates the wildfire burn severity based on the Monitoring Trends in Burn Severity program (Eidenshink et al. 2007). Black dots indicate paired aquatic invertebrate and riparian bird sampling (2015), and blue dots indicate only riparian bird sampling (2016)
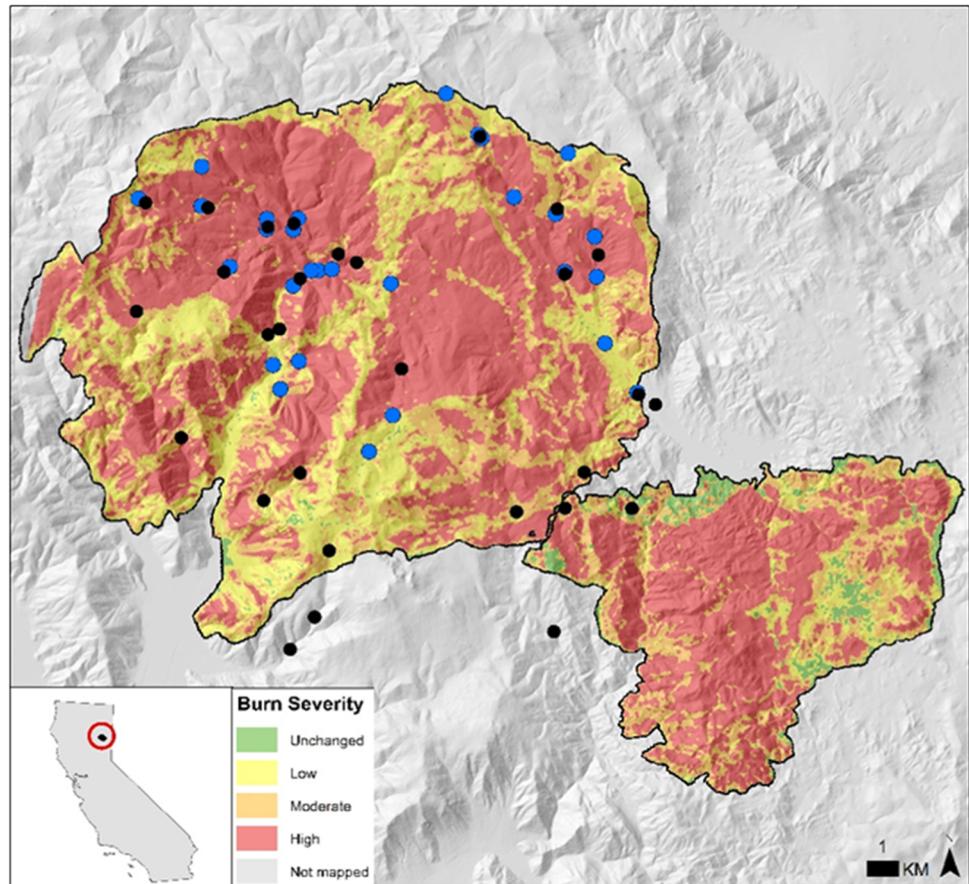
post-fire, limited studies have demonstrated the long-term effects of wildfire on soil nitrogen, carbon, and Hg concentrations and dynamics (see Banks et al. 2005, Patel et al. 2019). Additionally, sampling over 7 years post-fire allowed us to control for the short-term effects often observed 1–3 years after fires, such as increased MeHg in aquatic systems linked to mobilized soil Hg (see Kelly et al. 2006).

To examine the density and species composition of aquatic invertebrate taxa at each site, we collected 5 replicate aquatic invertebrate samples using a 929 cm^2 Surber sampler with 500 μm mesh. Individual replicate sample points were spaced 25 m apart perpendicular to the stream channel and samples were stored in 500 ml HDPE plastic bottles in 70% ethanol. Aquatic invertebrate samples were collected for MeHg analysis using dip nets at each site. Invertebrates were placed in plastic vials and stored on ice until they were frozen within 6 h. Water samples for sulfate and DOC analysis were collected in acid-washed bottles (triple rinsed with site water) at each site. Water samples were then filtered on-site using a 0.45 um filter into an amber glass bottle and preserved with hydrochloric acid. All samples collected in the field were kept cool on ice until long-term storage in a refrigerator.

Between May 27th and August 7th, 2015, and May 5th and August 4th, 2016, we sampled the riparian bird community at the previously identified sites (Fig. 1). Seventeen species of riparian birds were targeted with playback recordings of conspecific songs and captured using mist nets erected within 200 m of the stream channel. Each bird was aged and sexed following Pyle (1997). Birds were banded with an aluminum USGS leg band and whole blood samples were collected from the brachial vein before release at the capture site. In most cases riparian bird communities were sampled within 1 week (mean = 6.9 ± 1.4 d) of the aquatic community and water chemistry sampling (2015 only). All invertebrate and bird sampling was authorized under state (USGS; California: SC-4741, 8645) and federal permits (IBP; MB 22423).

Laboratory processing and analysis

Surber samples were rinsed through a 210 μm sieve to remove fine sediment and then subsampled using a 500 μm mesh Canton tray with random grids counted until ≥ 300 organisms were enumerated (Barbour et al. 1999). Samples containing fewer than the targeted 300 organisms were counted in their entirety. Invertebrates were identified (usually to family) using Merritt et al. (2008). Invertebrate families were then classified into three common foraging

guilds (predators, scrapers, grazers) and summed by foraging guild. To standardize our MeHg invertebrate analysis we selected representative families from each site that included predators (Dytiscidae, Perlidae, Rhyacophilidae), scrapers (Heptageniidae), and grazers (Limnephilidae). Invertebrates were sorted into families, rinsed with deionized water, and composited into glass vials for each family group. Invertebrate samples were dried for a minimum of 48 h at 50 °C. Samples were ground with either a glass rod in the vial or in a mortar and pestle to homogenize samples.

Mercury determination – aquatic invertebrates

We analyzed MeHg concentrations in all invertebrate samples following EPA method 1630 (U.S. Environmental Protection Agency, 2001) at the USGS Contaminant Ecology Research Lab in Corvallis, OR. We digested 2–10 mg of dried homogenate in 34 ml 30% nitric acid at 60 °C overnight (~15 h), ethylated with 1% sodium tetraethyl borate, then analyzed via cold-vapor atomic fluorescence spectrometry on a MERX-M (Brooks Rand Instruments, Seattle, Washington, USA) automated methylmercury analyzer. Invertebrates were analyzed for MeHg because ratios of MeHg:THg in invertebrates are less consistent than in other taxa (Tremblay et al. 1996). All Hg values are reported on a dry weigh basis (dw; unless stated otherwise), and quality assurance protocols including matrix blanks, duplicates, and spikes were used. Recoveries for calibration verification standards were $102.5 \pm 2.3\%$ ($n = 54$) for MeHg. Certified reference material recoveries were $98.5 \pm 2.2\%$ ($n = 45$) for MeHg. Absolute relative percent difference for duplicates averaged $4.43 \pm 1.7\%$ ($n = 24$). Matrix spike recoveries for MeHg averaged $108 \pm 3.2\%$ ($n = 14$).

Bird blood

We measured total mercury (THg) concentrations in blood because >95% of the Hg in bird blood is in the form of MeHg (Rimmer et al. 2005). Each sample was weighed onto a pre-weighed quartz filter and the wet weight was recorded before the sample was dried for a minimum of 48 h at 50 °C. Samples were returned to room temperature in a desiccator before dry weight was recorded. We analyzed all samples for THg following US EPA method 7473 (US EPA 2000) on a Milestone DMA-80 Direct Mercury Analyzer (Milestone Inc., Monroe, Connecticut, USA) or Nippon MA-3000 (Nippon Instruments Corporation, Tokyo, Japan). Quality assurance measures included analysis of two certified reference materials (either fish muscle tissue [DORM-4; National Research Council of Canada, Ottawa, Canada] or lobster hepatopancreas [TORT-3; National Research Council of Canada, Ottawa, Canada]), two system and

method blanks, and two duplicates per batch of 40 samples. Recoveries averaged $101.3 \pm 0.9\%$ ($n = 61$) and $103.1 \pm 1.4\%$ ($n = 44$) for certified reference materials and calibration checks, respectively. Absolute relative percent difference for duplicates averaged $10.4 \pm 4.4\%$ ($n = 13$). If MeHg or THg was detected in a sample, but below the (0.93–1.27 and 0.08–0.09 ng/g respectively) and ≥ the limit of detection (LOD; the lowest concentration in a sample that could be detected but not necessarily quantified as an exact concentration), we used the reported concentration rather than a ½ LOD which is commonly used but has no statistical underpinning (Zoffoli et al., 2013). Dry weight THg concentrations were converted back to wet weight (ww) using the individual moisture content.

Water chemistry

All water sample analysis was conducted at the Oregon State University Cooperative Chemical Analytical Laboratory, Corvallis, OR. Dissolved sulfate (SO_4; mg/L) was measured with a Dionex 1500 ion chromatograph, dissolved organic carbon (DOC; mg/L) was quantified on a Shimadzu TOC-VCSH combustion analyzer, and pH was measured using a ManTech PC-Titrate Auto Titrator System. Sulfate and DOC QAQC included method blanks and duplicates. Absolute relative percent difference for sulfate and DOC duplicates averaged $2.11 \pm 3.30\%$ ($n = 4$) and $0.01 \pm 0.01\%$ ($n = 3$), respectively.

Landscape variables – burn severity

We calculated burn severity using ArcMap 10.6.1 (ESRI, Redlands, California, USA) and the Monitoring Trends in Burn Severity (MTBS; U.S. Geological Survey, National Center for Earth Resources and Observation and Science and the USDA Forest Service, Geospatial Technology and Applications Center Eidenshink et al. 2007). The MTBS program maps the burn severity and extent of large fires (>1000 ha) across all lands of the United States (Eidenshink et al. 2007). Burn severity layers are created by first calculating spectral indices from pre- and post-fire satellite imagery that are sensitive to changes caused by fire. The two images are then subtracted showing the difference between them which is then thresholded into burn severity classes. MTBS classifies burn severity into five discrete classes: 1 (unburned/unchanged), 2 (low severity), 3 (moderate severity), 4 (high severity), and 5 (increased post-fire response). Using the classified severity rasters, we calculated the percentage of each stream catchment burned across the range of burn severity classes (1–5). Because burn severity differed within the stream catchments where we sampled aquatic invertebrates and riparian birds, we created a weighted average burn severity index for each

sampling site that accounted for the proportion of each catchment that fell in the above severity categories.

Statistical analyses

We first examined the influence of foraging guild on aquatic invertebrate MeHg concentrations and riparian bird THg concentrations. MeHg concentration was the response variable for aquatic invertebrates, foraging guild (predator, scraper, shredder) was the lone independent variable, and sampling site was a random effect. For riparian birds, we included individual bird THg concentrations as the response variable, foraging guild (insectivore, omnivore), foraging behavior (bark forager, flycatcher, ground forager, high canopy forager, low canopy forager; DeGraaf et al. 1985), and age class (hatch-year or adult) as independent variables, and year, site and species as random effects. We did not include the sex of birds in the model because 53% of the birds (largely hatch-year birds) sampled were not assigned sex. We natural log-transformed all Hg data across all models to improve the normality of the residuals and homogenize the variance structure.

Next, we tested the importance of weighted average burn severity and limnological variables on invertebrate and bird Hg concentrations. First, we generated overall least squares mean invertebrate MeHg or bird THg concentrations for each sampling site from models that included invertebrate MeHg or bird THg concentration as the responses, and site and trophic guilds as independent variables. Because aquatic invertebrates were only sampled in 2015, we constrained least squares mean THg concentrations for birds to those sampled in the same year. We then ran separate models for each benthic invertebrate group (predator, scraper, shredder) because of potential differences in feeding pathways, and we wanted to test the effects of landscape and limnological variables in each guild independently. We included burn severity as a main effect and stream chemistry (dissolved organic carbon, sulfate, and pH) as covariates. We also included a burn severity × DOC interaction to determine whether the relationship between DOC and invertebrate MeHg concentrations differed among burn severities and removed the interaction when results were not significant ($P > 0.0.05$). We tested for multicollinearity between stream chemistry variables prior to formal analysis, finding no evidence of multicollinearity (R^2s = 0.00–0.07).

Insectivorous and omnivorous birds are likely exposed to mercury through different feeding pathways, so we ran separate models for each foraging guild. Because MeHg concentrations in each invertebrate trophic guild were correlated (R^2s = 0.10–0.29) with the paired MeHg concentrations of the other guilds sampled from the same sites, we could not include multiple guilds as independent variables in the same models. Therefore, we generated overall least squares mean invertebrate MeHg concentration for

each sampling site from a model that included invertebrate MeHg concentrations as the response, and site and invertebrate trophic guild as independent variables. Our final bird THg models took the form of either least squares mean insectivorous or omnivorous bird THg as the response variable and burn severity, DOC, sulfate, and pH as independent variables. Additionally, we included site-specific total invertebrate density and overall invertebrate least squares mean MeHg concentration as independent variables to determine if prey availability and prey MeHg concentrations influenced the local riparian bird community THg exposure. We included all plausible interactions and removed interactions when not significant ($P > 0.05$). To facilitate interpretation of the interaction between DOC and fire severity on scraper MeHg concentrations, we plotted the conditional coefficients of the effect of either DOC or fire severity on scraper MeHg concentrations relative to the range of either fire severity or DOC using package *interplot* (Solt et al. 2022) in R, version 3.4.2 (R Core Team 2021). This approach plots the changes in the coefficient of one variable in a two-way interaction term conditional on the value of the other included variable (Solt et al. 2022).

Results

Across all sites, individual MeHg concentrations varied by 710-fold (range = 0.4–263 ng/g dw, $n = 344$). The highest concentrations were in predatory stoneflies (Perlidae; 39.8; range = 33.5–47.4 ng/g dw), which was more than twice as high as concentrations in shredding caddisfly of the Limnephilidae family (18.4; range = 15.5–21.8 ng/g dw; Fig. 2).

We captured 17 species of riparian birds and across all species, sites, and years, the geometric mean whole blood THg concentration was 39.8 ± 1.4 ng/g dw, and individual THg concentrations varied by 760-fold (range = 1.1–835 ng/g ww, $n = 412$). The highest concentrations were in western wood-peewees (*Contopus sordidulus*, 237; range = 199.1–283.3 ng/g ww), which were approximately 20-fold higher than those in lazuli buntings (*Passerina amoena*, 11.7; range = 7.6–18.0 ng/g ww; Fig. 3), which had the lowest concentrations.

Aquatic invertebrate MeHg

Aquatic macroinvertebrate MeHg concentrations differed among trophic guilds ($F_{2,315.5} = 84.60$, $P < 0.0001$) after accounting for site effects (Fig. 4). Predator, scraper, and shredder aquatic macroinvertebrates all differed from each other (Tukey HSD Ps = <0.001–0.001; Fig. 4). MeHg concentrations were highest in predators, and were 1.8 and 2.5-fold higher than in scrapers and shredders, respectively.
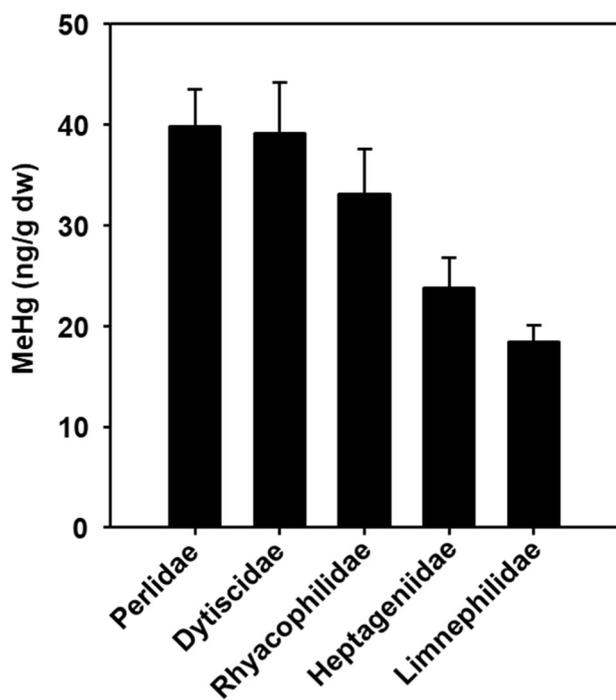
Fig. 2 Geometric mean aquatic invertebrate MeHg (ng/g dw) + geometric standard error concentrations by family from the Plumas National Forest, CA during 2015

Fig. 3 Geometric mean riparian bird THg (ng/g ww) + geometric standard error concentrations by species from the Plumas National Forest, CA during 2015 and 2016. Results include hatch year and adult (after hatch year) birds combined

Wildfire and biogeochemical effects on aquatic invertebrate MeHg

Variation in predator and shredder MeHg concentrations were not influenced by burn severity (predator: $F_{1,25} = 1.77$, $P = 0.19$; shredder: $F_{1,22} = 2.36$, $P = 0.14$), pH (predator: $F_{1,25} = 0.41$, $P = 0.53$; shredder: $F_{1,22} = 0.07$, $P = 0.79$), DOC (predator: $F_{1,25} = 2.06$, $P = 0.16$; shredder $F_{1,22} = 0.04$, $P = 0.84$), or sulfate (predator: $F_{1,25} = 0.13$, $P = 0.72$; shredder $F_{1,22} = 0.00$, $P = 0.96$), nor the burn severity × DOC interaction (predator: $F_{1,24} = 0.26$, $P = 0.6$; shredder: $F_{1,21} = 0.20$, $P = 0.66$). In contrast, scraper MeHg concentrations had a significant interaction between burn severity and DOC ($F_{1,19} = 4.48$, $P = 0.05$). To facilitate the interpretation of this interaction on scraper MeHg concentrations, we plotted the conditional slope coefficients for the effect of DOC concentrations on scraper invertebrate MeHg concentrations across the range of burn severity (Fig. 5). This illustrates that the magnitude and direction of the relationship between DOC concentrations and scraper MeHg concentrations change depending on burn severity. At the lowest burn severities (<2) there was a marginally positive relationship between DOC and scraper MeHg concentrations (95% CIs overlapped the 0-slope minimally), and as burn severity increased the relationship became neutral (95% CIs completely overlapped 0-slope line), and then was strongly negative (95% CIs do not overlap the 0-slope line) at the highest burn severities (>2.75; Fig. 5).

Fig. 4 Least squares mean (± SE) methylmercury (MeHg; ng/g dw) concentrations among the three primary aquatic invertebrate trophic guilds sampled during 2015 in the Plumas National Forest, CA, after accounting for site effects. Letters represent significant differences between groups

Riparian bird THg

Blood THg concentrations in riparian birds (ng/g ww) differed between foraging guilds ($F_{1,11.74} = 7.02$, $P = 0.02$), but not among foraging behaviors ($F_{4,10.08} = 2.88$, $P = 0.08$) or
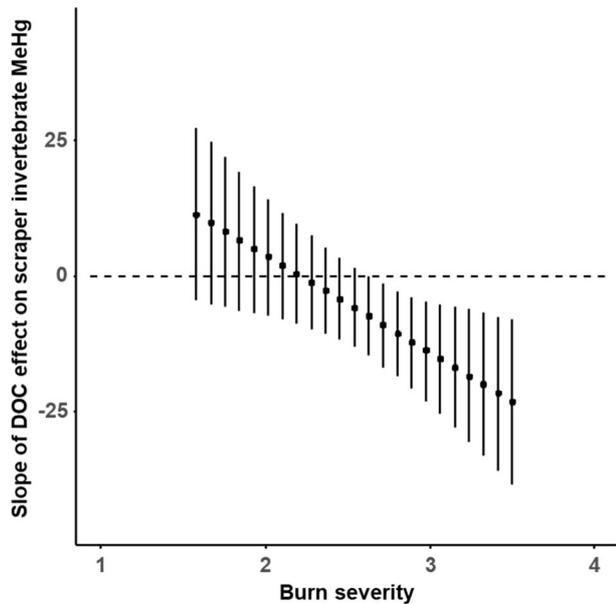
Fig. 5 Conditional effects of burn severity on the estimated coefficient of dissolved organic carbon (DOC mg/L) on scraper invertebrate MeHg (ng/g dw) concentrations in the Plumas National Forest during 2015, after accounting for pH and sulfate. The vertical lines indicate 95% confidence intervals around the estimated slope coefficient

Fig. 6 Least squares mean THg concentrations (ng/g ww) ± SE for the riparian foraging guilds in riparian bird THg in the Plumas National Forest, CA, after accounting for species, site, and year effects

age ($F_{1,375.60} = 0.16$, $P = 0.69$), after accounting for collection site, species, and year. Total Hg concentrations in insectivores were 2.6-fold higher than those in omnivores (Fig. 6).

Blood THg concentrations in insectivorous birds (ng/g ww) were positively correlated with DOC concentrations ($F_{1,20} = 4.33$, $P = 0.05$) and pH ($F_{1,20} = 6.55$, $P = 0.02$; Fig. 7A, B). However, both effects were the result of a single outlier that had a considerable amount of leverage and when those outliers were excluded from models these effects were no longer significant (Ps = 0.46 and 0.10, respectively). Blood THg concentrations in insectivorous birds were not influenced by burn severity ($F_{1,20} = 0.82$, $P = 0.38$), sulfate ($F_{1,20} = 0.74$, $P = 0.40$), invertebrate MeHg concentrations ($F_{1,20} = 0.00$, $P = 0.98$), or invertebrate density ($F_{1,20} = 2.42$, $P = 0.14$), and all interactions were not significant ($P > 0.31$ for all tests). In contrast, blood THg concentrations in omnivorous birds were negatively influenced by burn severity ($F_{1,19} = 8.00$, $P = 0.01$; Fig. 7C), but not correlated with DOC concentrations ($F_{1,19} = 2.15$, $P = 0.16$), sulfate ($F_{1,19} = 0.63$, $P = 0.44$), pH ($F_{1,19} = 2.24$, $P = 0.15$), invertebrate MeHg concentrations ($F_{1,19} = 2.04$, $P = 0.17$), or invertebrate densities ($F_{1,19} = 2.22$, $P = 0.10$) and all interactions were not significant ($P > 0.18$ for all tests).

Discussion

Across 29 riparian sites that ranged from no fire damage to high fire severity within the Plumas National Forest during

the Moonlight and Antelope Wildfires, we found benthic invertebrate MeHg concentrations varied by 710-fold and riparian bird THg concentrations varied by 760-fold. Aquatic invertebrate MeHg and riparian bird THg concentrations also differed by guild. Methylmercury concentrations in scraper invertebrates, but not predator or shredder invertebrates, were correlated with an interactive effect of DOC and burn severity, and we found strong relationships between riparian bird THg concentrations and drivers of abiotic MeHg cycling (burn severity).

Species, foraging guild, and foraging style effects on mercury concentrations

Foraging guilds for both aquatic invertebrates and riparian birds differed in Hg concentrations similar to previous studies (Riva-Murray et al. 2011, Jardine et al. 2012, Willacker et al. 2019). Not surprisingly, both predacious aquatic invertebrates and insectivorous riparian birds had the highest Hg concentrations in this study. Similarly, within the riparian bird foraging behaviors, birds in the insectivore foraging guild had the highest THg concentrations. Differences in diet-derived Hg concentrations have been demonstrated previously and are likely associated with biomagnification factors attributed to food web lengths (Ackerman et al. 2016, Knutsen and Varian-Ramos 2020) and reliance on aquatic derived prey (Cristol and Evers 2020, Jackson et al. 2021).

The highest Hg concentrations in a specific bird species were observed in the western wood-pewees (insectivore); geometric mean concentrations were 2.4-fold higher than the next species of riparian bird (house wren, low canopy forager insectivore). Although flycatcher bird species that
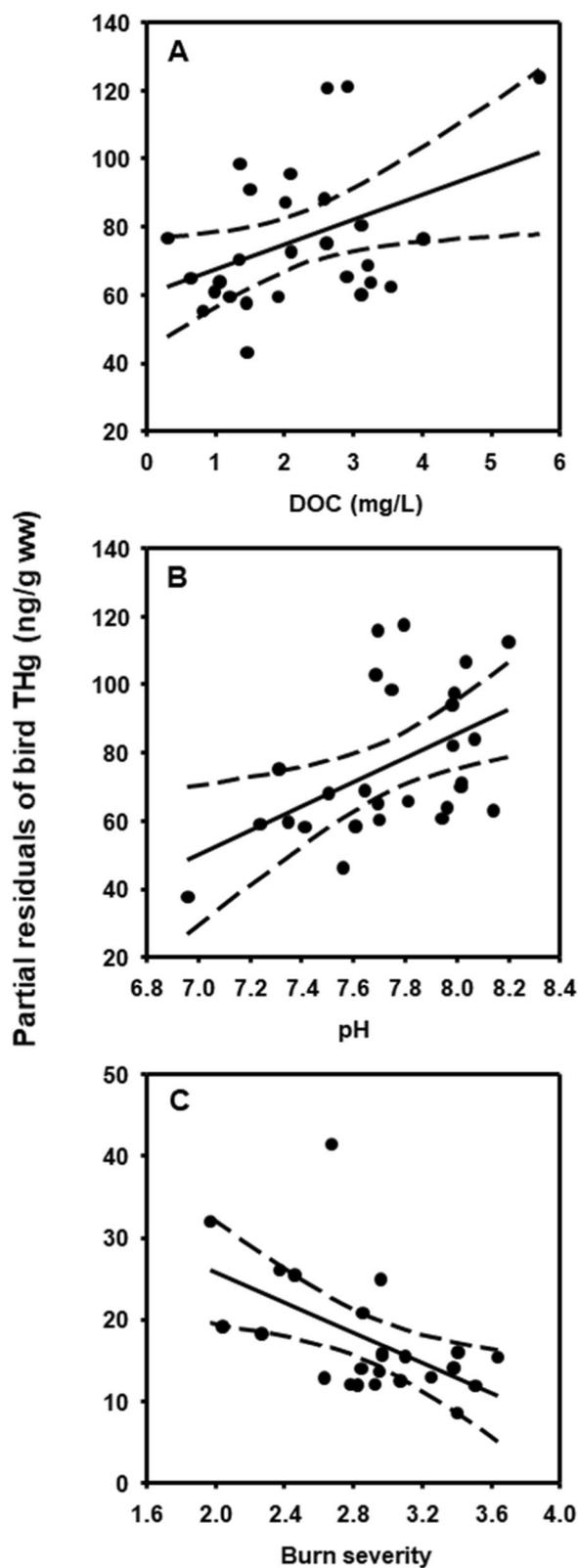
Fig. 7 Influence of **A** dissolved organic carbon (DOC mg/L) and **B** pH on insectivorous bird THg (ng/g ww) concentrations and **C** burn severity on omnivorous bird THg in the Plumas National Forest during 2015. The dashed lines indicate 95% confidence intervals

almost strictly rely on aquatic subsidies (e.g., emergent insects) had the highest Hg concentration as a group, there exists considerable variability even within this group. Western wood-pewee Hg concentrations were 2.5-fold higher than the next highest flycatcher species, the dusky flycatcher, suggesting that within a foraging guild, there are considerable differences in the reliance on aquatic prey (Jackson et al. 2021). The fact that western wood-pewee THg concentrations were elevated relative to all other species across all sites suggests the species is targeting invertebrate species that have elevated MeHg concentrations. Hg concentrations in western wood-pewees have been found to be some of the highest of all flycatchers in other studies of songbirds (Knutsen and Varian-Ramos 2020).

Site-specific biogeochemistry is an important determinant of MeHg exposure and concentrations in resident aquatic species (Snodgrass et al. 2000, Mitchell and Gilmour 2008, Eagles-Smith et al. 2016). Despite this, we found inconsistent relationships between MeHg concentrations in different aquatic macroinvertebrate guilds and stream chemical parameters. MeHg concentrations were linked to an interactive effect of DOC concentrations and burn severity (burn severity moderated the effect of DOC on MeHg concentrations) in scrapers, but not predators or shredders suggesting the taxa more trophically linked to in situ energetic pathways (aquatic versus aquatic and terrestrial) are also more responsive to the biogeochemical processes that influence MeHg cycling. Similarly, while research has shown that MeHg in water often increases with DOC, MeHg concentrations in stream invertebrates do not consistently increase (Chaves-Ulloa et al. 2016).

Changes in productivity and community structure at the base of stream food webs can also impact MeHg bioaccumulation (Ward et al. 2010, Willacker et al. 2019). While scrapers had intermediate MeHg concentrations, the MeHg consumed through feeding on epiphytic biofilms may be more strongly tied to local stream chemistry than that of grazers because of the high methylation rates that occur on epiphytes (Cleckner et al. 1999, Mauro et al. 2004, Desrosiers et al. 2006). Additionally, wildfires open up tree canopies, increasing periphyton growth that can enhance scraper populations and decrease MeHg concentrations (Ward et al. 2010). In contrast, shredders may be disconnected from local stream chemistry due to their feeding on terrestrial leaf material, whereas concentrations in predators are likely confounded by preying upon insects across guilds that derive their MeHg from both terrestrial and aquatic sources.

Similar to scraper aquatic macroinvertebrates, in riparian birds we found linkages to factors that influenced biogeochemical processes (e.g., burn severity) that impacted Hg concentrations in omnivorous birds. However, we did not find a correlation between stream invertebrate MeHg

concentrations and riparian insectivorous or omnivorous bird THg concentrations. The lack of correlation between aquatic invertebrate MeHg and riparian bird Hg concentrations suggests the prey source for these birds may not be strictly aquatic in origin. Previous studies have shown that other sources of terrestrial prey (e.g., spiders) can play an important role in influencing riparian bird Hg concentrations (Cristol et al. 2008, Townsend et al. 2014, Jackson et al. 2021). Additionally, potential linkages to aquatic food sources may be complicated by the fact that before hatching, these aquatic invertebrates themselves may have poor linkages to the local stream chemistry due to feeding on aquatic and terrestrial sources of food (Erdozain et al. 2019).

Burn severity has the potential to alter Hg in aquatic systems through several actions. Remarkably, nearly 8 years post-fire, we still observed evidence of burn severity influencing both scraper invertebrate MeHg and omnivorous bird THg concentrations. Previous studies have also demonstrated the negative effects of fire on stream taxa Hg concentrations, although in these studies the time between fire and sampling differed considerably (~2 years post-fire; Allen et al. 2005 and ~53 years post-fire: Bank et al. 2005). The mechanism(s) by which fire results in lower Hg is unclear but could be associated with removing Hg pools from adjacent terrestrial habitats (Shanely et al. 2005, Driscoll et al. 2013, Eagles-Smith et al. 2016, Webster et al., 2016), reducing carbon supply to the stream system that can facilitate Hg transport (Shanely et al. 2005), or decreasing biological uptake (Dutton and Fisher 2012). Similarly, increased burn severity reduces terrestrial Hg stores and available carbon required for terrestrial MeHg production (Mailman and Bodaly 2005), likely decreasing Hg availability to riparian birds via terrestrial invertebrates. Additionally, increased light availability associated with burned stream watersheds could alter instream productivity resulting in bloom dilution (Ward et al. 2010, Wagner et al. 2017). Alternatively, reduced THg concentrations in riparian birds may be more directly related to decreased MeHg availability in terrestrial habitats and subsequently lower Hg in terrestrial invertebrates consumed by riparian birds (Rimmer et al. 2005, Obrist et al. 2012). Repeated sampling of stream invertebrates, riparian birds, and stream chemistry variables across time after a fire may help disentangle some of the processes that drive these differing Hg patterns and correlations with local aquatic and riparian systems.

Wildfire regimes are rapidly changing due to changing climate patterns, extreme weather, and land use patterns (Rogers et al. 2020). The influences of these changes on Hg distribution/redistribution at landscape scales and potential effects at local scales remains uncertain. Our research adds to this understanding by demonstrating how aquatic invertebrate and riparian bird communities' Hg exposure can be affected by wildfire and the associated changes in the local aquatic environment. Additionally, we demonstrated the relationship between DOC and invertebrate and bird Hg concentrations and the interaction between wildfire burn intensity and DOC. These findings demonstrate that local water chemistry, adjacent landscape, and ecological factors all need to be considered when examining Hg accumulation and distribution with aquatic and riparian communities.

Data availability

The data are available at Herring et al. 2023.

Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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References

Ackerman JT, Eagles-Smith CA, Herzog MP, Hartman CA, Peterson SH, Evers DC, Jackson AK, Elliot JE, Vander Pol SS, Bryan CE (2016) Avian mercury exposure and toxicological risk across western North America: a synthesis. Sci Total Environ 568:749–769

Allen EW, Prepas EE, Gabos S, Strachan WMJ, Zhang WP (2005) Methylmercury concentrations in macroinvertebrates and fish from burned and undisturbed lakes on the boreal plain. Can J Fish Aquat Sci 62:1963–1977

Bank MS, Loftin CS, Jung RE (2005) Mercury bioaccumulation in northern two-lined salamanders from streams in the northeastern United States. Ecotoxicology 14:181–191

Barbour MT, Gerritsen J, Snyder BD, Stribling JB (1999) Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, 2nd edn. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

Barrow NJ, Cox VC (1992) The effects of pH and chloride concentration on mercury sorption II. by a soil. J Soil Sci 43:305–312

Biswas A, Blum JD, Klaue B, Keeler GJ (2007) Release of mercury from Rocky Mountain forest fires. Global Biogeochem Cycles 21(GB1002):1–13

Brown JA, Robertson BL, McDonald T (2015) Spatially balanced sampling: application to environmental survey. Procedia Environ Sci 27:6–9

Cal Fire (2023) Incidents | CAL FIRE. Accessed 14 Nov 2023

Chaves-Ulloa R, Taylor BW, Broadley HJ, Cottingham KL, Baer NA, Weathers KC, Ewing HA, Chen CY (2016) Dissolved organic carbon modulates mercury concentrations in insect subsidies from streams to terrestrial consumers. Ecol Appl 26:1771–1784

Chen CF, Ju YR, Lim YC, Chen CW, Wu CH, Lin YL, Dong CD (2020) Dry and wet seasonal variation of total mercury, inorganic mercury, and methylmercury formation in estuary and harbor sediments. J Environ Manage 253:109683

Cleckner LB, Gilmour CC, Hurley JP, Krabbenhoft DP (1999) Mercury methylation in periphyton of the Florida everglades. Limnol Oceanogr 44:1815–1825

Craig PJ, Moreton PA (1985) The role of speciation in mercury methylation in sediments and water. Environ Pollut Series B 10:141–158

Cristol DA, Brasso R, Condon AM, Fovargue RE, Friedman SL, Hallinger KK, Monroe AP, White AE (2008) The movement of aquatic mercury through terrestrial food webs. Science 320:335

Cristol DA, Evers DC (2020) The impact of mercury on North American songbirds: effects, trends, and predictive factors. Ecotoxicology 29:1107–1116

DeGraaf RM, Tilghman NG, Anderson SH (1985) Foraging guilds of North American birds. Environ Manag 9:493–536

Desrosier M, Planas D, Mucci A (2006) Mercury methylation in the eipilithon of Boreal Shield aquatic ecosystems. Environ Sci Technol 40:1540–1546

Driscoll CT, Mason RP, Chan HM, Jacob DJ, Pirrone N (2013) Mercury as a global pollutant: sources, pathways, and effects. Environ Sci Technol 47:4967–4983

Dutton J, Fisher NS (2012) Influence of humic acid on the uptake of aqueous metals by the killifish Fundulus heteroclitus. Environ Toxicol Chem 31:2225–2232

Eagles-Smith CA, Herring G, Johnson B, Graw R (2016) Conifer density within lake catchments predicts fish mercury concentrations in remote subalpine lakes. Environ Pollution 212:279–289

Eagles-Smith CA, Silbergeld EK, Basu N, Bustamante P, Diaz-Barriga F, Hopkins WA, Kidd KA, Nyland JF (2018) Modulators of mercury risk to wildlife and humans in the context of rapid global change. Ambio 47:170–197

Eidenshink J, Scwind B, Brewer K, Zhu Z, Quayle B, Howard S (2007) A project for monitoring trends in burn severity. Fire Ecology 3:3–21

Erdozain M, Kidd K, Kreutweiser D, Sibley P (2019) Increased reliance of stream macroinvertebrates on terrestrial food sources linked to forest management intensity. Ecol Appl 29:e01889

Friedli HR, Radke LF, Lu JY, Banic CM, Leatich WR, MacPherson JI (2003) Mercury emissions from burning of biomass from temperate North American forests: laboratory and airborne measurements. Atmos Environ 37:253–267

Grigal DF (2003) Mercury sequestration in forests and peatlands: a review. J Environ Quality 32:393–405

Herring G, Eagles-Smith CA, Tennant LB, Willacker JJ, Johnson M, Siegel RB, Polasik JS (2023) Mercury in bird blood and benthic aquatic invertebrates in Plumas National Forest, 2015–2016: U.S. Geological Survey data release, https://doi.org/10.5066/P9OZ0R9M

Hsu-Kim H, Eckley CS, Achá D, Feng X, Gilmour CC, Jonnson S, Mitchell CPJ (2018) Challenges and opportunities for managing aquatic mercury pollution in altered landscapes. Ambio 47:141–169

Jackson AK, Eagles-Smith CA, Robinson WD (2021) Differential reliance on aquatic prey subsidies influences mercury exposure in riparian arachnids and songbirds. Ecol Evol 11:7003–7017

Jardine TD, Kidd KA, Rasmussen JB (2012) Aquatic and terrestrial organic matter in the diet of stream consumers: implications for mercury bioaccumulation. Ecol Appl 22:843–855

Kelly EN, Schindler DW, St Louis VL, Donald DB, Vladicka KE (2006) Forest fire increases mercury accumulation by fishes via food web restructuring and increased mercury inputs. Proc Natl Acad Sci USA 103:19380–19385

Kim CS, Rytuba JJ, Brown Jr GE (2003) EXAFS study of mercury(II) sorption to Fe- and Al-(hydr)oxides. II. Effects of chloride and sulfate. J Colloids Inter Sci 270:9–20

Knutsen CJ, Varian-Ramos CW (2020) Explaining variation in Colorado songbird blood mercury using migratory behavior, foraging guild, and diet. Ecotoxicology 29:1268–1280

Lambertsson L, Nilsson M (2006) Organic material: the primary control on mercury methylation and ambient methylmercury concentrations in estuarine sediments. Environ Sci Tech 40:1822–1829

Mailman M, Bodaly RA (2005) Total mercury, methylmercury, and carbon in fresh and burned plants and soil in Northwestern Ontario. Environ Pollut 138:161–166

Mauro J, Guimarães J, Hintelmann H, Watras C, Haack E, Coelho-Souza S (2004) Mercury methylation in macrophytes, periphyton, and water—comparative studies with stable and radio-mercury additions. Anal Bioanal Chem 374:983–989

Mehrotra AS, Sedlak DL (2005) Decrease in net mercury methylation rates following iron amendment to anoxicwetland sediment slurries. Environ Sci Tech 39:2564–2570

Merrit RW, Cummins KW, Berg MB (2008) An introduction to the aquatic insects of North America, 4th edition. Kendall Hunt

Mitchell CPJ, Gilmour CC (2008) Methylmercury production in a Chesapeake Bay salt marsh. J Geophys Res Biogeosci 113:G00C04

NWCG [National Wildfire Coordinating Group] (2005) Glossary of wildland fire terminology. National interagency fire center. Boise, Idaho http://www.nwcg.gov/pms/pubs/pubs. Htm#PMS205. Accessed 20 May 2021

Obrist D, Johnson DW, Edmonds RL (2012) Effects of vegetation type on mercury concentrations and pools in two adjacent coniferous and deciduous forests. J Plant Nutr Soil Sci 175:68

Obrist D, Kirk JL, Zhang L, Sunderland EM, Jiskra M, Selin NE (2018) A review of global environmental mercury processes in response to human and natural perturbations: changes of emissions, climate, and land use. Ambio 47:116–140

Patel KF, Jakubowski MD, Fernandez IJ et al. (2019) Soil nitrogen and mercury dynamics seven decades after a fire disturbance: a case study at Acadia National Park. Water Air Soil Pollut 230:29

Pyle P (1997) Identification guide to north American birds Part 1: Columbidae to Ploceidae. 2nd edn. Slate Creek Press, Point Reyes Station, CA, USA

Rimmer CC, McFarland KP, Evers DC, Miller EK, Aubry Y, Busby D, Taylor RJ (2005) Mercury levels in Bicknell's thrush and other insectivorous passerine birds in montane forests of the northeastern United States and Canada. Ecotoxicology 14:223–240

Riva-Murray K, Chasar LC, Bradley PM, Burns DA, Brigham ME, Smith MJ, Abrahamsen TA (2011) Spatial patterns of mercury in macroinvertebrates and fishes from streams of two contrasting forested landscapes in the eastern United States. Ecotoxicology 20:1530–1542

R Core Team (2021) R: A language and environment for statistical computing. R Foundation for Statistical Computing. https://www.R-project.org/

Rogers BM, Balch JK, Goetz SJ, Lehmann CER, Turetsky M (2020) Focus on changing fire regimes: interactions with climate, ecosystems, and society. Environ Res Letter 15:030201

Shanely J, Kamman N, Clair TA, Chalmers AT (2005) Physical controls on total and methylmercury in streams and lakes of Northeastern USA. Ecotoxicology 14:125–134

Snodgrass J, Jagoe CH, Bryan Jr AL, Brant HA, Burger J (2000) Effects of trophic status and wetland morphology, hydroperiod,

and water chemistry on mercury concentrations in fish. Can J Fish Aquat Sci 57:171–180

Solt F, Hu Y, Kenkel B (2022) Package 'interplot'. Interplot: Plot the Effects of Variables in Interaction Terms (fhcrc.org)

Townsend JM, Driscoll CT, Rimmer CC, McFarland KP (2014) Avian, salamander, and forest floor mercury concentrations increase with elevation in a terrestrial ecosystem. Environ Toxicol Chem 33:208–215

Tremblay A, Lucotte M, Rheault I (1996) Methylmercury in a benthic food web of two hydroelectric reservoirs and a natural lake of northern Quebec (Canada). Water Air Soil Pollut 91:255–269

U.S. Environmental Protection Agency (2000) Mercury in solids and solutions by thermal decomposition, amalgamation, and atomic absorption spectrophotometry. Method 7473. Test methods for evaluating solid waste, physical/chemical methods SW 846, Update IVA; US Government Printing Office: Washington, DC

U.S. Environmental Protection Agency (2001) Methyl mercury in water by distillation, aqueous ethylation, purge and trap, and cold-vapor atomic fluorescence spectrometry. Office of Water and Office of Science and Technology, Washington, DC. Method 1630. EPA-821-R-01-020

Wagner K, Bengtsson MM, Findlay RH, Battin TL, Ulseth AJ (2017) High light intensity mediates a shift from allochthonous to autochthonous carbon use in phototrophic stream biofilms. J Geophys Res: Biogeo 122:1806–1820

Ward DM, Nislow KH, Folt CL (2010) Bioaccumulation syndrome: identifying factors that make some stream food webs prone to elevated mercury bioaccumulation. Annal New York Acad Sci 1195:62–83

Webster JP, Kane TJ, Obrist D, Ryan JN, Aiken GR (2016) Estimating mercury emissions resulting from wildlife in forests of the Western United States. Sci Total Environ 568:578–586

Wei X, Hayes DJ, Fernandez I (2021) Fire reduces riverine DOC concentration draining a watershed and alters post-fire DOC recovery patterns. Environ Res 16:024022

Wiener JG, Knights BC, Sandheinrich MB, Jeremiason JD, Brigham ME, Engstrom DR, Woodruff LG, Cannon WF, Balogh SJ (2006) Mercury in Soils, Lakes, and Fish in Voyageurs National Park (Minnesota): Importance of Atmospheric Deposition and Ecosystem Factors. Environ Sci Technol 40:6261–6268

Willacker JW, Eagles-Smith CA, Kowalski BM, Denehy RJ, Jackson AK, Adams EM, Evers DC, Eckley CS, Tate MT, Krabbenhoft DP (2019) Timber harvest alters mercury bioaccumulation and food web structure in headwater streams. Environ Pollut 253:636–645

Windham-Myers L, Marvin-Dipasquale M, Krabbenhoft DP, Agee JL, Cox MH, Heredia-Middleton P, Kakouros E (2009) Experimental removal of wetland emergent vegetation leads to decreased methylmercury production in surface sediment. J Geophys 114:G2

Zoffoli HJ, Varella CA, do Amaral-Sobrinho NM, Zonta E, Tolon-Becerra A (2013) Method of median semi-variance for the analysis of left-censored data: comparison with other techniques using environmental data. Chemosphere 93:1701–1709