Ecotoxicology (2014) 23:304–316
DOI 10.1007/s10646-013-1174-6

Mercury bioaccumulation in Southern Appalachian birds, assessed through feather concentrations

Rebecca Hylton Keller · Lingtian Xie · David B. Buchwalter · Kathleen E. Franzreb · Theodore R. Simons

Accepted: 31 December 2013 / Published online: 14 January 2014
© Springer Science+Business Media New York 2014

Abstract Mercury contamination in wildlife has rarely been studied in the Southern Appalachians despite high deposition rates in the region. From 2006 to 2008 we sampled feathers from 458 birds representing 32 species in the Southern Appalachians for total mercury and stable isotope δ^{15}N. Mercury concentrations (mean ± SE) averaged 0.46 ± 0.02 μg g^{-1} (range 0.01–3.74 μg g^{-1}). Twelve of 32 species had individuals (7 % of all birds sampled) with mercury concentrations higher than 1 μg g^{-1}. Mercury concentrations were 17 % higher in juveniles compared to adults ($n = 454$). In adults, invertivores has higher mercury levels compared to omnivores. Mercury was highest at low-elevation sites near water, however mercury was detected in all birds, including those in the high elevations (1,000–2,000 m). Relative trophic position, calculated from δ^{15}N, ranged from 2.13 to 4.87 across all birds. We fitted linear mixed-effects models to the data separately for juveniles and year-round resident adults. In adults, mercury concentrations were 2.4 times higher in invertivores compared to omnivores. Trophic position was the main effect explaining mercury levels in juveniles, with an estimated 0.18 ± 0.08 μg g^{-1} increase in feather mercury for each one unit rise in trophic position. Our research demonstrates that mercury is biomagnifying in birds within this terrestrial mountainous system, and further research is warranted for animals foraging at higher trophic levels, particularly those associated with aquatic environments downslope from montane areas receiving high mercury deposition.

Keywords Mercury · Songbirds · Appalachian Mountains · Nitrogen-15 stable isotope · Trophic position · Terrestrial

R. H. Keller (✉)
Appalachian Mountains Joint Venture, American Bird Conservancy, 1900 Kraft Drive, Suite 250, Blacksburg, VA 24061, USA
e-mail: bkeller@abcbirds.org

L. Xie
Institute of Applied Ecology, Chinese Academy of Sciences, 72 Wenhua Road, Shenyang, People's Republic of China

L. Xie · D. B. Buchwalter
Department of Environmental and Molecular Toxicology, North Carolina State University, Campus Box 7633, Raleigh, NC 27695-7633, USA

K. E. Franzreb
Department of Forestry, Wildlife and Fisheries, University of Tennessee, Knoxville, TN 37996, USA

T. R. Simons
USGS NC Cooperative Fish and Wildlife Research Unit, Department of Biology, North Carolina State University, Campus Box 7617, Raleigh, NC 27695-7617, USA

Introduction

Mercury is a widespread, persistent contaminant of human and wildlife populations around the world (Eisler 2006). Inorganic mercury released from natural and industrial sources disperses globally and enters aquatic systems via atmospheric deposition, where its methylated form becomes biologically active and highly toxic (Scheuhammer et al. 2009). Mercury biomagnifies with increasing trophic position through the aquatic food web (Chen et al. 2005), including into fish-eating birds (Evers et al. 2005; Cristol et al. 2008; Eagles-Smith and Ackerman 2009), amphibians (Bergeron et al. 2010), and mammals (Yates et al. 2005; Dietz et al. 2006) associated with water. Despite recent advances in understanding the mechanisms of mercury methylation, transfer and accumulation in

freshwater aquatic systems (Evers and Clair 2005; Driscoll et al. 2007; Jardine et al. 2012), little is known about the uptake and transfer of mercury by organisms in terrestrial systems (Seewagen 2010).

Upland forest soils are natural sinks for atmospherically deposited mercury because the majority of mercury binds to organic and mineral soil particles (Hall and St. Louis 2004). Mercury is concentrated primarily in the upper soil horizons (Kim et al. 1997), and up to 60 % of mercury that reaches lakes originates from the terrestrial watershed (Krabbenhoft and Babiarz 1992; Griegal 2002). Mercury loading is significantly higher in coniferous compared to deciduous habitats, and two to five times higher in mountainous areas in the northeastern United States compared to nearby low-elevation areas (Lawson 1999; Miller et al. 2005). Consumption of contaminated food is the primary risk of mercury exposure to terrestrial vertebrates (Scheuhammer et al. 2012).

Birds are at a particularly high risk of mercury toxicity because many species occupy high trophic levels, are long-lived, and are vulnerable to neurological and reproductive impacts from elevated mercury levels (Burger 1993; Evers et al. 2005). The effects of mercury contamination in birds have been studied primarily in large piscivorous species such as common loons (*Gavia immer*, Evers et al. 2008), wading birds (Frederick et al. 2004, Frederick et al. 2011), bald eagles (*Haliaeetus leucocephalus*) and belted kingfishers (*Megaceryle alcyon*, Evers et al. 2005). Recently, there has been increased concern for mercury bioaccumulation in passerines (Morrissey et al. 2005; Shriver et al. 2006; Brasso and Cristol 2008; Cristol et al. 2008; Condon and Cristol 2009; Hawley et al. 2009; Wada et al. 2009; Jackson et al. 2011; Seewagen 2013; Warner et al. 2012), although studies of songbirds in strictly terrestrial systems remain limited (Rimmer et al. 2005, 2010; Seewagen 2010; Townsend et al. 2013).

Predicting a species' susceptibility to mercury toxicity based on its foraging guild (e.g., frugivore, invertivore) is a simple first step to identifying species at greatest risk of mercury contamination in the terrestrial environment (Rimmer et al. 2005; Osborne et al. 2011). Further, analyzing stable isotopes can precisely calculate a species' relative dietary trophic position, as higher stable ^{15}N isotope [3–4 ‰ (per mil)] is found in predators compared to their prey (Vander Zanden and Rasmussen 1999; Hobson and Bairlein 2003; Pearson et al. 2003; Vitz and Rodewald 2012). Trophic position based on isotopic signatures has been successfully used to predict mercury concentrations in seabirds (Atwell et al. 1998) loons (Burgess and Hobson 2006), and dippers (Morrissey et al. 2004, 2010).

Birds primarily reduce body burdens of mercury during feather molt as mercury has a high affinity for keratin, although adult females can also depurate mercury during egg production (Crewther et al. 1965). Feather mercury levels of juveniles reflect short-term, site-specific blood mercury concentrations (Bearhop et al. 2000; Condon and Cristol 2009). However, feather mercury levels of adults can reflect a combination of site-specific dietary uptake of mercury as well as chronic body burdens of mercury that have been remobilized from contaminated muscle tissue (Evers et al. 2005).

Mercury is a serious concern in the Southern Appalachians due to high rates of deposition on the landscape (National Atmospheric Deposition Program 2007), yet there has been no targeted mercury research on birds in this region. The primary objectives of this research were (1) to determine if mercury is bioaccumulating in high-elevation terrestrial birds in the Southern Appalachians; (2) to determine which species and individuals (e.g., juvenile vs. adult) are most at risk for elevated levels; (3) to test for a relationship between mercury bioaccumulation and trophic position using stable isotopes; and (4) to determine if environmental factors (e.g., vegetation, moisture, distance to water) are driving observed patterns of mercury loading. We predicted that feather mercury would be higher in (1) adults compared to juveniles due to chronic exposure, (2) birds that feed at higher trophic levels (i.e., those with higher stable nitrogen isotope ratios (δ^{15}N), and (3) invertivores compared to omnivores. Also, in the high elevations, we predicted higher mercury levels in (4) birds along coniferous ridge tops associated with increased atmospheric deposition compared to deciduous or mixed-deciduous, protected slopes (Weathers et al. 2000, 2006).

Methods

Study area

In 2002, year-round mercury deposition network collectors were installed in two locations in Great Smoky Mountains National Park (Fig. 1). Wet mercury deposition was estimated by the National Atmospheric Deposition Program as 15.5, 9.9 and 11.4 μg m^{-2} in 2006, 2007 and 2008, respectively. Elevated mercury levels in the park are attributed to both nearby coal-fired plants as well as more distant sources (Keeler et al. 2006), although Valente et al. (2007) did not find atmospheric mercury levels significantly higher than global trends in a nearby mid-elevation (813 m) site just west of Great Smoky Mountains National Park.

We sampled birds from multiple elevations and habitats, but focused primarily on birds breeding in the high-elevation (>1,500 m) Southern Appalachian red spruce (*Picea rubens*)-Fraser fir (*Abies fraseri)* forests within Great Smoky Mountains National Park (35°36′N, 83°25′W, 520,000 acres) and three additional high-elevation sites

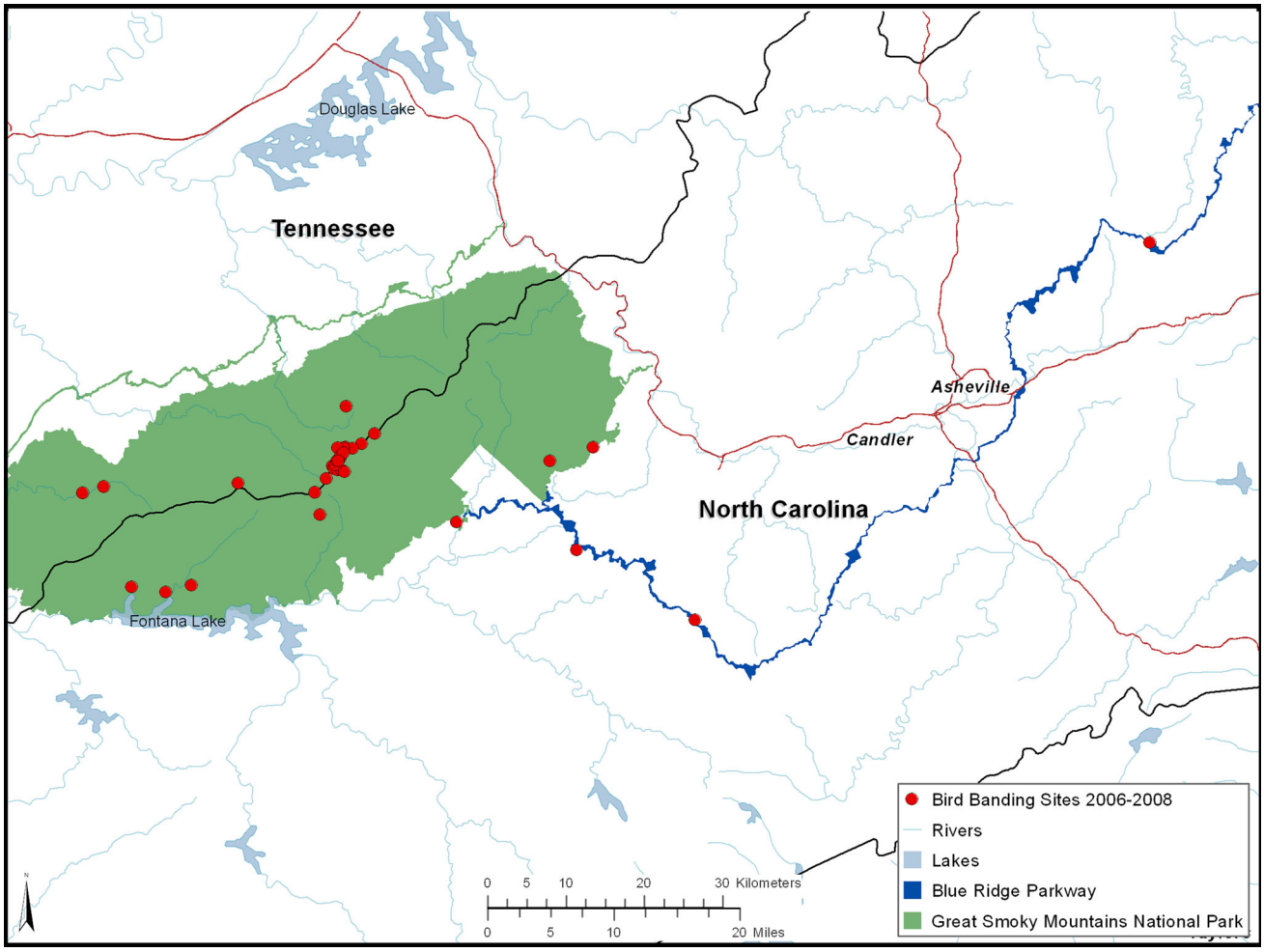

Fig. 1 Mist-netting sites ($n = 32$) for birds whose feathers were collected for mercury and δ^{15}N analysis in the Southern Appalachians within Great Smoky Mountains National Park and the Blue Ridge Parkway

along the southern portion of the Blue Ridge Parkway: the Richland Balsam summit (35°22′N, 82°99′W), Waterrock Knob (35°28′N 82°08′W) and the Black Rock Trailhead near Mt. Mitchell State Park (35°43′N, 82°16′W, Fig. 1). The red spruce–Fraser fir forest ecosystem is in jeopardy due to the synergistic effects of heavy fir loss by invasive balsam wooly adelgid beetles (*Adelges piceae*), foliar damage by high levels of ozone, and the direct and indirect effects of high levels of acid precipitation (Shortle and Smith 1988; Nodvin et al. 1995; McLaughlin and Wimmer 1999). Within the ecotone, red spruce dominates at lower elevations (<1,400 m), Fraser fir dominates at the highest elevations (>1,600 m) and the two codominate at mid-elevations (1,400–1,600 m). Approximately 74 % of the endangered red spruce-Fraser fir habitat of the eastern United States is located within Great Smoky Mountains National Park (National Park Service Air Resources Division 2002).

We collected feathers from 5 low-elevation (420–616 m) and 26 high-elevation (1,449–1,986 m) sites (Table 1). Study sites were established at least 200 m apart to reduce spatial autocorrelation, and within 100 m of roads or trails for accessibility. Birds were opportunistically captured using passive mist netting. The number of nets and effort at each site varied depending on the primary purpose of the site. For example, sampling at MAPS Stations (U.S. North American Bird Conservation Initiative Monitoring Subcommittee 2007) involved up to 20 nets and 8 personnel over 6 h, while at other sites as few as 5 nets were tended by a single person for 4 h. Due to these differences in capture methodology, capture data does not reliably indicate the impact of Hg on species composition or abundance across sites.

Feather mercury analysis

We banded each captured bird with an individually numbered USGS aluminum band, and recorded its species, age, and sex (Pyle 1997). We collected both S1 secondary flight feathers ($n = 458$ birds) from a total of 36 species across

Table 1 Summary of adult year-round resident birds captured in Great Smoky Mountains National Park and along the southern Blue Ridge Parkway from 2006 to 2008, including species common name, scientific name, year-round foraging guild, and mean and range of feather total mercury values

Species	Latin name	Foraging guild	*N*	Feather THg (μg g^{-1}, fw)	
				Mean ± SE	Range
American crow	*Corvus brachyrhynchos*	*Omnivore*	1	0.09	
Belted kingfisher	*Megaceryle alcyon*	*Piscivore*	1	0.99	
Black-capped chickadee	*Poecile atricapillus*	*Insectivore*	6	0.44 ± 0.10	0.26–0.99
Brown creeper	*Certhia americana*	*Insectivore*	4	0.68 ± 0.11	0.12–1.11
Carolina wren	*Thryothorus ludovicianus*	*Insectivore*	5	0.97 ± 0.25	0.27–3.74
Golden-crowned kinglet	*Regulus satrapa*	*Insectivore*	27	0.62 ± 0.06	0.09–2.05
Slate-colored junco[a]	*Junco hyemalis carolinensis*	*Omnivore*	143	0.40 ± 0.02	0.02–1.74
Song sparrow	*Melospiza melodia*	*Omnivore*	4	0.42 ± 0.06	0.07–1.16
Winter wren[a]	*Ttoglodytes hiemalis*	*Insectivore*	6	0.84 ± 0.10	0.38–1.55

[a] Indicates that this species is known to be a partial altitudinal migrant during harsh winter conditions in the Southern Appalachians, although information is lacking for most species

31 sites in Great Smoky Mountains National Park and along the Blue Ridge Parkway from 2006 to 2008. Flight feathers for juvenile birds are grown during the nestling and fledging periods (Pyle 1997) and thus represent the mercury levels near the nest. Mercury levels in songbird feathers vary depending on the order in which they are grown (Condon and Cristol 2009). Although molting sequence can vary across species, we sampled S1 secondaries as they are one of the first flight feathers grown/molted in most passerines; secondaries are typically grown on the breeding grounds prior to migrating for most species (Pyle 1997). However, to be conservative, we sampled feathers only from juveniles or year-round resident adults (Alsop 1991), not adults of migratory species. It should be noted that year-round residents may become altitudinal migrants during the coldest months. Feather samples were stored in clean, dry paper envelopes prior to mercury analysis at the North Carolina State University Department of Environmental and Molecular Toxicology using cold vapor atomic fluorescence spectroscopy (USEPA method 1631 revision E with a modified digestion procedure). Feathers were not freeze-dried or altered prior to mercury analysis (Lasorsa et al. 2012).

An S1 feather from both (2006) or one (2007, 2008) wing from each bird were fully homogenized and digested in 1.5 mL of 18 M H_2SO_4 (Leeman Labs Inc., West Chester, Pennsylvania, USA) and 4.5 mL of ultrapure 16 M HNO_3 (OmniTrace Ultra, Merck, Darmstadt, Germany) in 50 mL Teflon digestion vessels using a microwave heating system (MarsXress, CEM Inc, Mathews, North Carolina, USA). Samples were then subjected to BrCl oxidation (overnight) and $SnCl_2$ reduction. Samples were diluted with 0.2 % HCl for THg analysis by flow-injection cold-vapor atomic fluorescence spectrophotometry (CVAFS) with a Leeman Laboratories Hydro AF Gold plus analyzer (Leeman Labs Inc, USA).

The accuracy of THg determination was evaluated by analysis of certified standard reference material from the National Institute of Standards and Technology (NIST Mussel 2976), method blanks (acid), spiked and replicate samples and calibration standards. Our measured concentration (mean ± SD) of THg in NIST Mussel 2976 was 61.0 ± 4.0 ng g^{-1} dry weight ($n = 20$, certified value: 61 ± 3.6 ng g^{-1}, total recovery 100.1 ± 6.6 %). The average THg in method blanks was 0.14 ng L^{-1}. Mercury concentrations in all samples exceeded our method detection limit of 1.3 ng g^{-1} dry weight, calculated as three times the standard deviation of the method blank and divided by the average sample mass.

Stable isotope analysis

We analyzed nitrogen-15 stable isotopes ($\delta^{15}N$) in feather and caterpillar samples to evaluate each bird's relative trophic position in the food web, as the nitrogen isotopic signature of prey items influences the isotopic signature of a consumer's tissues at the time they were grown (Vander Zanden and Rasmussen 1999; Pearson et al. 2003). In 2007 and 2008, we analyzed $\delta^{15}N$ (Post et al. 2000) in the remaining S1 secondary feather that was not tested for THg from 236 birds for at the Alaska Stable Isotope Facility at the University of Alaska Fairbank's Water & Environmental Research Center. As birds tend to grow secondary feathers on both wings simultaneously and in sequence, we were able to match an individual bird's THg levels in a S1 feather from one wing with $\delta^{15}N$ levels in the feather from the opposite wing. We cleaned each feather using a dilute detergent solution, followed by a 2:1 chloroform:methanol

solvent, and three deionized water rinses (Paritte and Kelly 2009). After feathers were dried, we removed the terminus of each feather (0.200–0.500 mg) and placed the samples in tin capsules. These tin capsules were then closed and placed in a Carlo Erba NC2500 elemental analyzer autosampler, where they were combusted. Stable isotope ratios were obtained using continuous-flow isotope ratio mass spectrometry using a Delta V mass spectrometer interfaced with a Costech ESC 4010 elemental analyzer. The combustion reactor consisted of a reaction tube packed with chromium oxide and silver/cobalt oxide, and the reduction tube was packed with reduced copper wire. Stable isotope ratios were reported in δ notation as parts per thousand (‰) deviation from the international standard atmospheric N_2 ($\delta^{15}N_{AIR}$). Quality control involved analyzing tin capsule blanks every 20 samples and laboratory working standards (Peptone No. P-7750, meat based protein, Sigma Chemical Company Lot #76f-0300) every 10 samples ($n = 44$, $\delta^{15}N = 7.0 \pm 0.2$ ‰, N = 7.0 ‰).

We determined the trophic position (isotopic discrimination factor, $\Delta^{15}N$) of each bird by analyzing the isotopic differences in δ values between species, using the following formula from Post (2002):

$$\text{Trophic position} = 2 + \left(\delta^{15}N_{\text{secondaryconsumer}} - \delta^{15}N_{\text{herbivore}}\right)/3.4\,‰$$

with birds as the secondary consumer and caterpillars as the herbivores. The 3.4 ‰ represents the average $\delta^{15}N$ enrichment expected for each trophic level in vertebrates (Post 2002); this value is consistent with feeding trials in yellow-rumped warblers (*Dendroica coronata*), a North American songbird (Pearson et al. 2003). We chose caterpillars as the primary consumer to provide the baseline N isotope signatures for each because caterpillars are primary consumers that are commonly fed to nestling birds. The caterpillars collected from banding sites in 2008 to serve as baseline $\delta^{15}N$ values (Vander Zanden and Rasmussen 1996) were inadvertently destroyed in fall 2009; therefore we recollected another set in late summer of 2010. We assumed that relative differences in isotopes between sites would remain consistent across years. We collected 8–9 caterpillars at each of the 33 sites by beating tree branches and other vegetation over a 1 × 1 m white sheet for 3 min. Caterpillar samples from a given site were pooled, frozen, dried at 75 °C for 48 h in a drying oven, and homogeneously ground into a fine powder using a mortar and pestle. The resulting material was analyzed to produce a single herbivore $\delta^{15}N$ value for each site.

Environmental variables

We classified each site's vegetation as conifer, deciduous, or mixed based on its dominant overstory composition using the Nature Serve vegetation model for Great Smoky Mountains National Park (White et al. 2003). We included Topographic Relative Moisture Index (TRMI, Parker 1982) as a covariate in our models (see below), which describes the relative moisture at a site, and is a summed scalar index of four landscape elements derived from a Digital Elevation Model: relative slope position, gradient, shape, and aspect. We chose TRMI as a covariate in our models because this parameter represents moisture patterns and is highly correlated with predicted lead deposition (Weathers et al. 2006). Patterns of lead deposition are likely similar to the deposition patterns of other atmospheric particulate matter, including mercury, and lead deposition in the soil indicates long-term deposition patterns, including wet, dry, and cloud deposition (Weathers et al. 1995, 2000). Landforms for the high-elevation sites were classified as steep slope (including ridge tops, upper slopes, slope crests, and steep slopes), flat summit, or side slope (including side slopes and coves) as calculated by USGS Southeast GAP Analysis Project (www.basic.ncsu.edu/segap).

Statistical analysis

We fitted separate linear mixed-effects models to adult and juvenile feather mercury data (Zuur et al. 2013). Adults and juveniles were analyzed separately due to the large difference in species composition (adults: $n = 7$; juveniles: $n = 23$) between the groups. The independent variables of interest were foraging guild (adults only; De Graaf et al. 1985), sex, TRMI, trophic position, vegetation, and two interaction terms: TRMI * trophic position and vegetation * trophic position. Foraging guild was not included in the juvenile models as invertebrates are the primary food for all nestling bird species in this study (Poole 2005). Three random variables (species, site, and distance to water) were considered for all models, however both species and distance to water held no explanatory power for mercury levels in adult birds, and were omitted in the final models. We also fitted a linear mixed-effects model to high-elevation birds only (adults and juveniles combined) to examine environmental factors associated with increased mercury bioaccumulation. The independent variables included in this model were elevation, landform, TRMI, vegetation, and distance to water (i.e., stream), with species and site as random effects.

We used the lmer function in package lme4 (Bates et al. 2013) in R (version 3.0.2, R Core Team 2013) to find the best-fitting models. Starting from a full (maximum) model, we selected and compared models using Akaike information criteria (AIC; Akaike 1973). We considered models with $\Delta AIC_c < 2$ to be competing (Burnham and Anderson 2002). Variable importance was assessed by summing Akaike weights across all models incorporating the same variable. We report the weighted mean of the coefficients

and associated standard error for the top models for both adults and juveniles. The use of *p*-values for significance testing is not recommended with these models (Bates et al. 2013). Adult feather Hg concentrations were log-transformed prior to analysis, and the residuals of each final model were visually inspected for normality (Zuur et al. 2009). For direct comparisons we used linear regression to examine the influence of species, age, sex, trophic position, and foraging guild, on feather mercury levels. All predictor variables were tested for collinearity and were found to be independent (correlation coefficients <0.20).

Results

We sampled feathers for mercury analysis from 197 adults of 9 species that were year-round residents (Table 1) and 259 juveniles of 32 species (both resident and migratory; Table 2) in the Southern Appalachians. Slate-colored juncos (*Junco hyemalis carolinensis*) accounted for 56 % of all captures. Mercury was detected in all individuals sampled. All feathers combined had an average mercury load (mean ± SE) of 0.46 ± 0.02 μg g^{-1} (range 0.01–3.74 μg g^{-1}, $n = 458$, Table 3). In a direct comparison, feather mercury concentrations were 17 % higher in juveniles compared to adults when all species were combined ($t = 2.45$, $n = 454$, $p = 0.01$). In adults, on average females showed 20 % higher mercury levels compared to males, however this difference was not significant given the high variability among individuals (($t = 1.87$, $n = 191$, $p = 0.06$).

Twelve of 32 species had individuals (7 % of all birds sampled) with mercury levels >1 μg g^{-1} (Fig. 2): Carolina wren (31 % of individuals), winter wren (*Troglodytes troglodytes*, 33 %), veery (*Catharus fuscescens*, 50 %), slate-colored junco (4 %), Canada warbler (*Wilsonia canadensis*, 8 %), indigo bunting (*Passerina cyanea,* 25 %), eastern towhee (*Pipilo erythrophthlamus*, 33 %), golden-crowned kinglet (*Regulus satrapa*, 10 %), song sparrow (*Melospiza melodia*, 4 %), black-throated blue warbler (*Dendroica caerulescens*, 6 %), and brown creeper (*Certhia americana*, 12 %). Two species in separate genera had feather mercury >2 μg g^{-1}: a Carolina wren (15 %) and a golden-crowned kinglet (3 %). Only one individual exceeded 3 μg g^{-1} mercury: a Carolina wren (3.74 μg g^{-1}, 8 %) captured at the mouth of Eagle Creek where it spills into Lake Fontana, a reservoir that forms part of the southern border of Great Smoky Mountains National Park and the northern border of Nantahala National Forest. Of these 11 species, 7 are invertivores, while 4 are omnivores (Table 1).

Mercury levels varied widely among sites, with the highest levels recorded at Eagle Creek and Great Smoky Mountains Institute at Tremont, both low-elevation sites near water (Table 3). In addition to the Carolina wren, only one other bird, a juvenile hooded warbler (*Setophaga citrina*) with a mercury concentration of 0.91 μg g^{-1} was captured at Eagle Creek, resulting in a mean of 2.33 μg g^{-1} mercury for the site. Birds at all other sites averaged less than 1 μg g^{-1} mercury. The next highest average mercury levels were located at Great Smoky Mountains Institute at Tremont, which hosts a MAPS Station along the Middle Prong, Little River. Here, Carolina wrens and Louisiana waterthrushes had the highest mercury concentrations and are both invertivores that forage along this river.

The raw stable isotope δ^{15}N values for birds in our study ranged from +1.82 to 11.5 ‰, with a mean of +4.50 ± 0.29 ‰, while the δ^{15}N values for caterpillars ranged from −2.55 to +2.78 ‰, with a mean of +0.18 ± 0.01 ‰. Relative trophic position of the birds ranged from 2.13 to 4.87, with a mean of 3.27 ± 0.21. There was no difference in trophic position between males and females for adult birds ($t = 0.40$, $n = 124$, $p = 0.69$). Trophic position was not significantly different between juveniles and adults ($t = 1.853$, $n = 235$, $p = 0.07$).

In adults, invertivores (0.76 ± 0.03 μg g^{-1}, $n = 48$) had significantly higher mercury concentrations compared to omnivores (0.32 ± 0.02 μg g^{-1}, $n = 127$; $t = 7.80$, $p < 0.0001$, $R^2 = 0.23$). We have only one data point for a piscivore (belted kingfisher) whose feather mercury concentration was 0.99 μg g^{-1}, which is intermediate among the species reported. However, as the only representative of this foraging guild it was not included in the modeling analysis.

For adults, we compared a total of 16 models. Trophic position (100 % of the variable weight), foraging guild (94 %), vegetation (52 %), the interaction between trophic position and vegetation (32 %), and sex (10 %) were the main effects in the top three models explaining adult mercury levels, with these models carrying 70 % of the model weights (Table 4). Using weighted averages for the fixed effects estimates whose 95 % confidence intervals (CI) did not cross zero, on average omnivores saw a 93 % (CI −118 to −67 %) decrease in mercury compared to invertivores and a 104 % (CI 1–207 %) increase in mercury in deciduous forest compared to coniferous forest. Site, the only random effect retained in the adult model, explained only 3 % of the total random effect variance of the model.

For juveniles, we compared a total of 10 models. There was only one competing model for juveniles, which included trophic position as a main effect, and site, distance to water, and species as random effects (Table 4). This model carried 57 % of the model weights, and estimated a 0.18 ± 0.08 μg g^{-1} increase in feather mercury for each

Table 2 Summary of juvenile birds captured in Great Smoky Mountains National Park and along the southern Blue Ridge Parkway from 2006 to 2008, including species common name, scientific name, and mean and range of feather total mercury values

Species	Latin name	*N*	Feather THg (μg g^{-1}, fw)	
			Mean ± SE	Range
American robin	*Turdus migratorius*	1	0.19	
Barn swallow	*Hirundo rustica*	1	0.59	
Black-and-white warbler	*Miniotilta varia*	2	0.35 ± 0.14	0.21–0.49
Blackburnian warbler	*Setophaga fusca*	1	0.58	
Black-throated blue warbler	*Setophaga caerulescens*	16	0.43 ± 0.05	0.24–1.04
Black-throated green warbler	*Setophaga virens*	1	0.12	
Blue-gray gnatcatcher	*Polioptila caerulea*	2	0.74 ± 0.02	0.72–0.76
Brown creeper	*Certhia americana*	4	0.68 ± 0.11	0.12–1.11
Canada warbler	*Cardellina canadensis*	12	0.61 ± 0.08	0.22–1.27
Carolina chickadee	*Poecile carolinensis*	4	0.12 ± 0.07	0.05–0.38
Carolina wren	*Thryothorus ludovicianus*	8	0.97 ± 0.25	0.27–3.74
Chestnut-sided warbler	*Setophaga pensylvanica*	10	0.21 ± 0.04	0.06–0.05
Common yellowthroat	*Geothlypis trichas*	1	0.24	
Downy woodpecker	*Picoides pubescens*	2	0.21 ± 0.02	0.19–0.22
Eastern meadowlark	*Sturnella magna*	1	0.12	
Eastern phoebe	*Sayornis phoebe*	9	0.40 ± 0.07	0.19–0.87
Eastern towhee	*Pipilo erythrophthalmus*	3	0.88 ± 0.27	0.55–1.41
Field sparrow	*Spizella pusilla*	1	0.17	
Golden-crowned kinglet	*Regulus satrapa*	10	0.62 ± 0.06	0.09–2.05
Gray catbird	*Dumetella carolinensis*	5	0.31 ± 0.06	0.12–0.45
Hooded warbler	*Setophaga citrina*	3	0.72 ± 0.13	0.46–0.91
House finch	*Carpodacus mexicanus*	1	0.01	
Indigo bunting	*Passerina cyanea*	4	0.51 ± 0.31	0.05–1.42
Least flycatcher	*Empidonax minimus*	1	0.60	
Louisiana waterthrush	*Parkesia motacilla*	3	0.76 ± 0.11	0.55–0.93
Ovenbird	*Seiurus aurocapilla*	9	0.33 ± 0.03	0.22–0.46
Rose-breasted grosbeak	*Pheucticus ludovicianus*	2	0.25 ± 0.09	0.16–0.34
Slate-colored junco	*Junco hyemalis carolinensis*	114	0.40 ± 0.02	0.02–1.74
Song sparrow	*Melospiza melodia*	21	0.42 ± 0.06	0.07–1.16
Veery	*Catharus fuscescens*	2	0.95 ± 0.51	0.44–1.46
Winter wren	*Troglodytes hiemalis*	6	0.84 ± 0.10	0.39–1.54
Yellow warbler	*Setophaga petechia*	1	0.12	
Yellow-breasted chat	*Icteria virens*	1	0.78	

one unit rise in trophic position. Of the random effects, species, distance to water, and site accounted for 39, 5, and 9 % of the total random effect variance of the model, respectively.

We compared a total of 16 models of environmental variables for high elevations birds only (Table 4). Elevation (100 % of the variable weight), TRMI (23 %), and distance to water (24 %) were the main effects in the top three models explaining mercury levels, with these models carrying 76 % of the model weights. Using weighted averages none of the environmental fixed effects estimates had 95 % CI which did not cross zero. Site and species, the only random effects retained in the environmental model, explained 4 and 32 % of the total random effect variance of the model, respectively.

Discussion

This study has provided the first direct evidence of mercury contamination to wildlife in Great Smoky Mountains National Park and the southern portion of the Blue Ridge Parkway. Our results indicate that high-elevation terrestrial songbirds in the Southern Appalachians are accumulating

Table 3 Mean (± SE) feather mercury values (μg g^{-1}, fw) by location for songbirds captured in Great Smoky Mountains National Park and along the southern Blue Ridge Parkway from 2006 to 2008

Site	Latitude	Longitude	Distance to water (m)	Elevation (m)	Life stage			
					Adult	Juvenile	Unknown	Combined
GSMI	35.4772	83.6894	26	420	1.11 ± 0.15 (4)	0.64 ± 0.14 (5)	–	0.85 ± 0.10 (9)
HACR	35.4736	83.7267	6	521	–	0.38 ± 0.09 (3)	–	0.38 ± 0.09 (3)
EACR	35.4843	83.7745	63	524	3.74 (1)	0.91 (1)	–	2.33 ± 1.42 (2)
CACO	35.5978	83.8258	13	528	–	0.54 ± 0.09 (11)	–	0.54 ± 0.09 (11)
LUFT	35.5094	83.3050	150	616	0.20 ± 0.09 (3)	0.39 ± 0.06 (15)	–	0.36 ± 0.05 (18)
P03	35.6106	83.4581	43	1449	0.30 (1)	–	–	0.30 (1)
PUKN	35.5864	83.0672	611	1452	0.29 ± 0.12 (6)	0.39 ± 0.03 (92)	–	0.38 ± 0.03 (98)
OCOV	35.6078	83.4380	286	1508	0.31 ± 0.08 (14)	0.71 ± 0.20 (5)	–	0.41 ± 0.09 (19)
P02	35.6100	83.4506	33	1,540	0.59 ± 0.28 (2)	–	–	0.59 ± 0.28 (2)
POGA	35.5633	83.1622	102	1,568	0.27 ± 0.10 (11)	0.42 ± 0.06 (17)	–	0.36 ± 0.05 (28)
RIBA08	35.7152	82.2744	482	1,569	0.48 ± 0.08 (14)	0.43 ± 0.10 (9)	0.90 (1)	0.48 ± 0.06 (24)
BKRT	35.6099	83.4472	259	1,580	0.46 ± 0.19 (5)	0.27 ± 0.05 (3)	–	0.39 ± 0.12 (8)
INGA	35.6112	83.4237	278	1,614	0.57 ± 0.09 (9)	0.72 ± 0.06 (34)	–	0.69 ± 0.05 (43)
NFGA	35.5828	83.4536	469	1,614	0.52 ± 0.08 (10)	0.49 ± 0.09 (7)	–	0.51 ± 0.06 (17)
P51	35.5850	83.4572	423	1,633	0.32 ± 0.02 (3)	–	–	0.32 ± 0.02 (3)
RIBA	35.6044	83.4511	309	1,653	0.30 ± 0.10 (13)	0.12 ± 0.04 (2)	–	0.28 ± 0.09 (15)
P50	35.5858	83.4622	294	1,662	0.23 ± 0.06 (3)	–	–	0.23 ± 0.06 (3)
P60	35.5383	83.4956	478	1,672	0.15 ± 0.02 (2)	–	–	0.15 ± 0.02 (2)
P49	35.5956	83.4599	276	1,697	0.21 ± 0.04 (3)	–	–	0.21 ± 0.04 (3)
P30	35.5961	83.4619	270	1,719	0.44 ± 0.11 (8)	0.42 ± 0.06 (9)	–	0.43 ± 0.06 (17)
SPFI	35.5981	83.4575	391	1,720	0.65 ± 0.17 (10)	0.42 (1)	–	0.63 ± 0.15 (11)
P57	35.6011	83.4556	557	1,724	0.33 ± 0.01 (4)	0.80 ± 0.15 (8)	–	0.65 ± 0.12 (12)
P58	35.5774	83.4804	326	1,732	0.52 ± 0.09 (3)	–	–	0.52 ± 0.09 (3)
P59	35.4594	83.1414	189	1,738	0.32 ± 0.05 (3)	–	–	0.32 ± 0.05 (3)
COGA	35.5872	83.4669	222	1,748	0.23 ± 0.05 (4)	0.24 ± 0.04 (2)	–	0.23 ± 0.03 (6)
WAKN	35.6210	83.4035	559	1,758	0.27 ± 0.04 (2)	0.32 ± 0.04 (5)	–	0.30 ± 0.03 (7)
P48	35.5909	83.4692	281	1,762	0.31 ± 0.12 (2)	–	–	0.31 ± 0.12 (2)
SWHE	35.3606	82.9874	625	1,774	0.37 ± 0.10 (9)	0.40 ± 0.06 (9)	0.59 (1)	0.40 ± 0.05 (19)
FORK	35.6564	83.4389	795	1,792	0.48 ± 0.09 (14)	–	–	0.48 ± 0.09 (14)
MTLE	35.3673	82.9901	817	1,918	0.31 ± 0.04 (12)	0.53 ± 0.06 (13)	–	0.42 ± 0.04 (25)
CLDO	35.5642	83.4986	573	1,986	0.46 ± 0.05 (22)	0.61 ± 0.13 (8)	–	0.50 ± 0.05 (30)
Total					0.43 ± 0.2 (197)	0.48 ± 0.02 (259)	0.75 ± 0.16 (2)	0.46 ± 0.02 (458)

Sample sizes are in parentheses

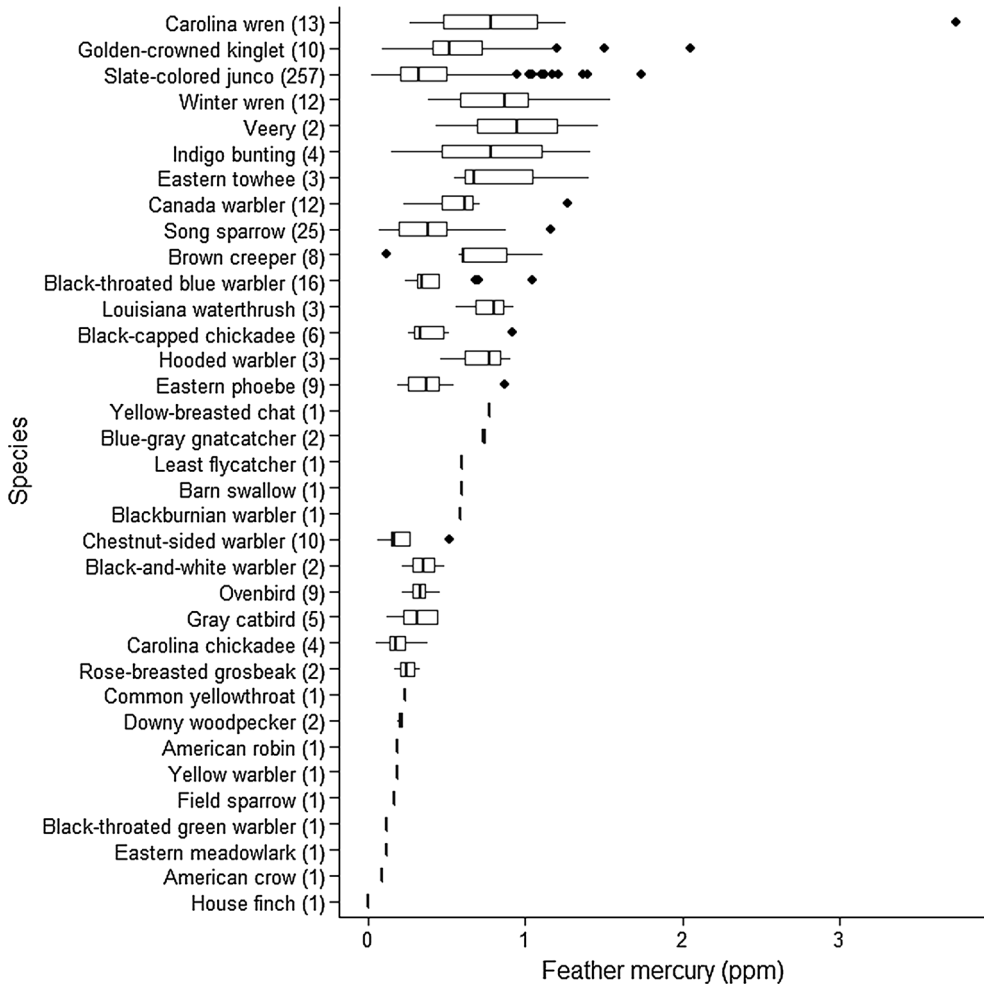

Fig. 2 Box-and-whisker plots of feather total mercury (THg) levels across all species (sample sizes). *Box outlines* indicate the 1st and 3rd quartiles, the *thicker central line* indicates the median, whiskers represent 95 % confidence intervals, and outliers are plotted as *black dots*. Small sample sizes for many species preclude these data from being representative of the species. Species are ordered by maximum mercury concentrations

mercury at concentrations similar to those documented in the northeastern U.S. (Rimmer et al. 2005). We found detectible mercury levels in every bird ($n = 458$) that we sampled (420–2,000 m elevation), with feather mercury concentrations higher than 1 μg g^{-1} in 7 % of individuals. Mercury concentrations varied across species, and much of this variation is likely related to foraging behavior. In adults, mercury concentrations were 2.4 times higher in invertivores compared to omnivores. These results are consistent with results from Rimmer et al. (2005, 2010) and Osborne et al. (2012).

We predicted that adults would have higher mercury levels compared to juveniles due to chronic body burdens, however mercury concentrations were 17 % higher in juveniles compared to adults. Mercury tends to increase with age (5–10 X higher in adults) in piscivorous birds (Evers et al. 2005), however this pattern has been poorly studied in terrestrial birds. Our results were consistent with those of Warner et al. (2012), who found that mercury concentrations in juvenile seaside (*Ammadramous maritimus*) and coastal plain swamp sparrow (*Melospiza geogiana nigrescens*) were higher than in adults. Warner et al. (2012) suggested that selective provisioning of high-protein food to nestlings or higher growth rates of nestlings, and subsequent higher food intake per body size, might explain these differences. Once the high growth period of the nestling phase is over, birds might able to reduce body burdens through feather depuration. Additionally, many of the species in our study switch to a more omnivorous diet during the non-breeding season (Poole 2005), which would yield lower mercury levels due to feeding at lower trophic levels.

We did not detect differences in mercury concentrations between adult males and females, suggesting that females did not depurate large amounts of mercury during egg-laying as seen in other species (Robinson et al. 2012). Our results are consistent with other studies which have found no effect of sex on blood mercury levels in belted kingfishers (Evers et al. 2005), tree swallows (Hallinger et al. 2011), and saltmarsh and Nelson's sharp-tailed sparrows (Shriver et al. 2006, Lane and Evers 2007).

Foraging guild was the strongest predictor of adult mercury concentrations. The average mercury concentration of

Table 4 Top and competing mixed linear-effects models used to explain feather mercury levels in adult and juvenile birds in the Southern Appalachians

Model	ΔAIC	AIC weight
Adults		
Guild + trophic position + (Site)	0.00	0.29
Guild + vegetation + trophic position + trophic position vegetation + (site)	0.45	0.23
Guild + vegetation + trophic position + (site)	0.87	0.19
Juveniles		
Trophic position + (site) + (distance to water) + (species)	0.00	0.57
High-elevation environmental factors model		
Elevation + (site) + (species)	0.00	0.29
Elevation + distance to water + (site) + (species)	0.31	0.24
Elevation + TRMI + (site) + (species)	0.47	0.23
Elevation + TRMI + distance to water + (site) + (species)	1.68	0.12

Fixed effects are listed first in the models, followed by random variables in parentheses. We present the difference in Akaike Information Criteria (AIC) values between the *i*th model and the lowest AIC value (ΔAIC) and Akaike weights for the set of models considered in the model selection process. A model with a lower ΔAIC and a higher AIC weight relative to the other models means that the given model is better at explaining the observed variability in our data. The number of parameters (typically denoted *K*) in the mixed effects models is difficult to calculate, and is therefore not reported here

adult invertivores was approximately 0.8 μg g^{-1} higher than that of adult omnivores, and maximum levels detected were approximately three times higher in adult invertivores compared to omnivores. This pattern suggests that mercury is biomagnifying in terrestrial songbirds as trophic level increases, which is supported by research in montane forests in the northeastern U.S. (Rimmer et al. 2010). Direct comparisons of our findings with those of Rimmer et al. (2010) are not possible, as they measured blood mercury, which tends to be much lower than feather mercury (Evers et al. 2005).

Stable isotope analysis has become increasingly common in avian ecology (Inger and Bearhop 2008; Morrissey et al. 2005, 2010). However, to our knowledge, this was the first study to link trophic position, based on δ^{15}N stable isotope values, to mercury concentrations in terrestrial birds. Relative trophic position of the birds ranged from 2.13 to 4.87, with a mean of 3.27. For comparison, a trophic position value of 1.00 would indicate an herbivore, and a value of 5.00 would indicate an individual that forages five steps up the food chain on average. Trophic position was the strongest predictor of mercury concentrations in juvenile birds. Trophic position also accounted for 100 % of the variable weight in the top adult models. We did not include foraging guild as a factor in the juvenile models as all nestlings in this study are likely fed a primarily invertivorous diet. Surprisingly, foraging guild was not highly correlated with trophic position in adults. This may reflect the differences between year-round diet and a largely invertivorous diet during the breeding season when insects are abundant. The inclusion of foraging guild in the top models for adults may better represent chronic mercury burdens compared to the short-term information on diet found in feathers grown during the breeding season. Among aquatic birds, Burgess and Hobson (2006) successfully used δ^{15}N stable isotopes to demonstrate that adult common loons had higher mercury levels and occupied a higher trophic level than juvenile loons in Canadian lakes, however there have been mixed results in seabird studies (Atwell et al. 1998; Bearhop et al. 2000; Bond and Diamond 2009).

Studying mercury accumulation in birds in the high elevations (1,000–2,000 m) was the primary focus of this research, however we also report mercury values for birds ($n = 43$, 9 %) captured at the lower elevations (420–620 m) of the Southern Appalachians. Birds captured at low-elevation sites associated with water bodies (i.e., rivers and lakes) showed the highest mercury levels, as might be expected (Evers et al. 2005). Of these, a Carolina wren captured by Lake Fontana had the highest mercury level recorded in this study. The propensity of Carolina wrens to consume spiders, which feed at higher trophic levels, likely explain the higher mercury concentrations observed compared to other species (Jackson et al. 2011). The lakeside capture location also suggests that mercury could be further magnifying through the aquatic food web, such as by eating prey items which spend their larval stage in the aquatic system before emerging as adults. We recommend further sampling of mercury in birds and other wildlife that feed at higher trophic levels in the Lake Fontana area, as lakes can act as reservoirs for mercury and support sulfate-reducing bacteria which converts inorganic mercury to the biologically toxic methylated form (Gilmour et al. 1992, Evers et al. 2005).

Although there are limited data for effects levels in terrestrial songbirds, Jackson et al. (2011) found that Carolina wrens showed 10, 20 and 30 % reductions in nesting success when mercury concentrations in body feathers were 2.4, 3.4, and 4.5 μg g^{-1}, respectively. Similarly, 10 and 20 % reductions in nesting success were seen at the respective effects concentrations of 3.0 and 4.7 μg g^{-1} in Carolina wren tail feathers. We examined mercury concentrations in a different feather type, secondary flight feathers, making direct comparisons difficult. However, if we extrapolate effects levels from the Jackson et al. (2011)

study, only one individual, an adult female Carolina wren, in our study had feather mercury concentrations (3.74 μg g^{-1}) that are likely associated with an estimated 10–25 % reduction in nesting success. Reductions in nesting success have also been associated with high mercury exposure in other avian species, including tree swallows (*Tachycineta bicolor*) (Brasso and Cristol 2008, Hallinger et al. 2011) and white ibises (*Eudocimus albus*, Frederick and Jayasena 2012).

Mean mercury concentrations were twice as high at some high elevation sites compared to other nearby high elevation sites. For this reason, we attempted to explain variation in mercury concentrations at high elevation sites using environmental variables such as vegetation, distance to water, moisture (TRMI), and land form (ridges versus protected slopes). Elevation, distance to water, and TRMI were in the top models, however none of these variables were significant at the 95 % confidence interval level. Our prediction that vegetation would be an important factor, with birds in coniferous habitats exhibiting higher mercury levels, was not supported. However, in the models looking at only adult birds, mercury levels were, on average, 104 % higher in deciduous forests compared to coniferous forests, although there was large variation in this estimate. The association of mercury levels increasing with increasing TRMI may be related to a combination of increased deposition rates, as well as increasingly wet soil and foliage surface conditions which would support the conversion of THg into bioavailable methylmercury (Miller et al. 2005). Similarly, Guigueno et al. (2012) found that THg levels in osprey (*Pandion haliaetus*) chicks in Canadian alpine lakes strongly increased with increasing local Hg deposition rates.

Summary and conclusions

In summary, our research demonstrated that mercury is biomagnifying in terrestrial birds across all elevations in the Southern Appalachians, and that stable isotopes are a useful tool for linking mercury to trophic position in songbirds. We suggest future studies use a combination of diet information (foraging guild plus trophic position, or evaluating trophic position at multiple times throughout the year) to better understand the influence of diet on mercury concentrations in adult songbirds. While we were able to document mercury concentrations in all birds captured, the lack of adverse effects levels research across multiple songbird species precludes evaluation of the effect of mercury on bird populations in the Southern Appalachians. Further mercury research is warranted for organisms which feed at higher trophic levels in this region, particularly those associated with low-elevation aquatic environments. In particular, low-lying water bodies of the Southern Appalachians that receive heavy drainage from nearby high elevations, such as Lake Fontana, NC, deserve closer scrutiny to determine levels of mercury contamination and potential hazards to wildlife populations. Understanding which watersheds and/or habitats are most likely to experience bioaccumulation of mercury will strengthen our knowledge of mechanisms involved in mercury contamination in the Southern Appalachians and other montane systems. The Southern Appalachians also experience high acidic deposition rates, and this acidic environment likely enhances methylation rates (Xun et al. 1987; Harmon et al. 2005), making the adverse effects of air pollution on wildlife populations synergistic and complex.

Acknowledgments This study was part of the Ph.D. dissertation of R. H. Keller at North Carolina State University and was financially supported by the US Geological Survey (Grant # 2005-1147 1568WO123), National Science Foundation (Award # 0910355), Appalachian Highlands Research Grants (AHRG2006-Keller), N.C. Beautiful, Burroughs Wellcome Fund, North Carolina State Graduate School, and the Park Flight Migratory Bird Program. We thank M. Brooks, M. Gutierrez, J. Thieme, C. Vanbianchi, T. Owens, R. Partida Lara, P. Elizondo, J. Runnels, C. Crider, T. Widen, P. Super, J. Fortner, L. Griggs, G. Tucker, E. Darling, and J. Harrold for assistance in the field. We also thank Drs. J. Collazo, K. Weathers and K. Pollock for serving on the graduate committee of R. H. Keller, the N.C. Gap Analysis Program for assistance with data layers, and K. Kidd, C. Rimmer and two anonymous reviewers for improving this manuscript. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Ethical standards All research conducted for this publication complied with the current laws and ethical standards of the United States of America.

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

References

Akaike H (1973) Information theory as an extension of the maximum likelihood principle. In: Petrov BN, Csaki F (eds) Second International Symposium on Information Theory. Akademiai Kiado, Budapest, pp 267–281

Alsop FJ Jr (1991) Birds of the smokies. Great Smoky Mountains Association, Gatlinburg

Atwell L, Hobson KA, Welch HE (1998) Biomagnification and bioaccumulation of mercury in an arctic marine food web: insights from stable nitrogen isotope analysis. Can J Fish Aquat Sci 55:1114–1121

Bates D, Maechler M, Bolker B, Walker S (2013) lme4: Linear mixed-effects models using Eigen and S4. R package version 1.0-5. http://CRAN.R-project.org/package=lme4. Accessed 15 Oct 2013

Bearhop S, Ruxton GD, Furness RW (2000) The dynamics of mercury in the blood and feathers of great skuas. Environ Toxicol Chem 19:1638–1643

Bergeron CM, Bodinof CM, Unrine JM, Hopkins WA (2010) Mercury accumulation along a contamination gradient and nondestructive indices of bioaccumulation in amphibians. Environ Toxicol Chem 29:980–988

Bond AL, Diamond AW (2009) Mercury concentrations in seabird tissues from Machias Seal Island, New Brunswick, Canada. Sci Total Environ 407:4340–4347

Brasso RL, Cristol DA (2008) Effects of mercury exposure on the reproductive success of tree swallows (*Tachycinea bicolor*). Ecotoxicology 17:133–141

Burger J (1993) Metals in avian feathers: bioindicators of environmental pollution. Rev Environ Toxicol 1993:203–311

Burgess NM, Hobson KA (2006) Bioaccumulation of mercury in yellow perch (*Perca flavescens*) and common loons (*Gavia immer*) in relation to lake chemistry in Atlantic Canada. Hydrobiologia 567:275–282

Burnham KP, Anderson DR (2002) Model selection and multimodel inference: a practical information-theoretic approach, 2nd edn. Springer-Verlag, New York

Chen CY, Stemberger RS, Kamman NC, Mayes BM, Folt CL (2005) Patterns of Hg bioaccumulation and transfer in aquatic food webs across multi-lake studies in the northeast U.S. Ecotoxicology 14:135–147

Condon AM, Cristol DA (2009) Feather growth influences blood mercury levels of young songbirds. Environ Toxicol Chem 28:395–401

Crewther WG, Fraser RD, Lennox FG, Lindley H (1965) The chemistry of keratins. In: Anifinsen CB, Anson ML, Edsoll JT, Richards FM (eds) Advances in protein chemistry. Academic Press, New York, pp 191–303

Cristol DA, Brasso RL, Condon AM, Fovargue RE, Friedman SL, Hallinger KK, Monroe AP, White AE (2008) The movement of aquatic mercury through terrestrial food webs. Science 329:335

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

Dietz R, Riget F, Born EW, Sonne C, Grandjean P, Kirkegaard M, Olsen MT, Asmund G, Renoni A, Baagøe H, Andreasen C (2006) Trends in mercury in hair of Greenlandic polar bears (Ursus maritimus) during 1892–2001. Environ Sci Technol 40:1120–1125

Driscoll CT, Han Y, Chen CY, Evers DC, Fallon Lambert K, Holsen TM, Kamman NC, Munson RK (2007) Mercury contamination in forest and freshwater ecosystems in the Northeastern United States. BioSci 57:17–28

Eagles-Smith C, Ackerman JT (2009) Rapid changes in small fish mercury concnetrations in estuarine wetlands: implications for wildlife risk and monitoring programs. Environ Sci Technol 43:8658–8664

Eisler R (2006) Mercury hazards to living organisms. Taylor and Francis Publishers, London

Evers DC, Clair TA (2005) Mercury in North-eastern North America: a synthesis of existing databases. Ecotoxicology 14:7–13

Evers DC, Burgess NM, Champoux L, Hoskins B, Major A, Goodale WM, Taylor RJ, Poppenga R, Daigle T (2005) Patterns and interpretation of mercury exposure in freshwater avian communities in northeastern North America. Ecotoxicology 14:193–221

Evers DC, Savoy LJ, DeSorbo CR, Yates DE, Haso W, Taylor KM, Siegel LS, Cooley HH Jr, Bank MS, Major A, Muey K, Mower B, Vogel HS, Schoch N, Pokras M, Goodale MW, Fair J (2008) Adverse effects from environmental mercury loads on breeding common loons. Ecotoxicology 17:69–81

Frederick PC, Jayasena N (2012) Altered pairing behavior and reproductive success in white ibises exposed to environmentally relevant concentrations of methylmercury. Proc R Soc Lond Ser B 278:1851–1857

Frederick PC, Hylton B, Heath JA, Spalding MG (2004) A historical record of mercury contamination in Southern Florida (USA) as inferred from avian feather tissue. Environ Toxicol Chem 23:1474–1478

Frederick PC, Campbell A, Jayasena N, Borkhataria R (2011) Survival of White Ibises (*Eudocimus albus*) in response to chronic experimental methylmercury exposure. Ecotoxicology 20:358–364

Gilmour CC, Henry EA, Mitchell R (1992) Sulfate stimulation and mercury methylation in freshwater sediments. Environ Sci Technol 26:2281–2287

Griegal DR (2002) Inputs and outputs of mercury from terrestrial watersheds: a review. Environ Rev 10:1–39

Guigueno MF, Elliott KH, Levac J, Wayland M, Elliott JE (2012) Differential exposure of alpine ospreys to mercury: melting glaciers, hydrology or deposition patterns? Environ Int 40:24–32

Hall BD, St Louis VL (2004) Methylmercury and total mercury in plant litter decomposing in upland forests and flooded landscapes. Environ Sci Technol 38:5010–5021

Hallinger KK, Cornell KL, Brasso RL, Cristol DA (2011) Mercury exposure and survival in free-living tree swallows (*Tachycineta bicolor*). Ecotoxicology 20:39–46

Harmon SM, King JK, Gladden JB, Chandler GT, Newman LA (2005) Mercury body burdens in *Gambusia holbrooki* and *Erimyzon sucetta* in a wetland mesocosm amended with sulfate. Chemosphere 59:227–233

Hawley DM, Hallinger KK, Cristol DA (2009) Compromised immune competence in free-living tree swallows exposed to mercury. Ecotoxicology 18:499–503

Hobson KA, Bairlein F (2003) Isotopic fractionation and turnover in captive Garden Warblers (*Sylvia borin*): implications for delineating dietary and migratory associations in wild passerines. Can J Zool 81:1630–1635

Inger R, Bearhop S (2008) Applications of stable isotope analysis to avian ecology. Ibis 150:447–461

Jackson AK, Evers DC, Etterson MA, Condon AM, Folsom SB, Detweiler J, Schmerfeld J, Cristol DA (2011) Mercury exposure affects the reproductive success of a free-living terrestrial songbird, the Carolina Wren (*Thryothorus ludovicianus*). Auk 128:759–769

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

Keeler GJ, Landis MS, Norris GS, Christianson EM, Dvonch JT (2006) Source of mercury wet deposition in Eastern Ohio, USA. Environ Sci Technol 40:5874–5881

Kim KH, Hansom PJ, Barnett MO, Lindberg SE (1997) Biogeochemistry of mercury in the air-soil-plant system. In: Sigel A, Sigel H (eds) Metal ions in biological systems, vol 34., Mercury and its Effects on Environment and BiologyMarcel Dekker, New York, pp 185–212

Krabbenhoft DP, Babiarz C (1992) The role of groundwater transport in aquatic mercury cycling. Water Resour Res 28:3119–3128

Lane OP, Evers DC (2007) Methylmercury availability in New England estuaries as indicated by Saltmarsh Sharp-tailed Sparrow, 2004–2006. BioDiversity Research Institute. BRI Report 2007–14

Lasorsa BK, Gill GA, Horvat M (2012) Analytical methods for measuring mercury in water, sediment and biota. In: Bank MS (ed) Mercury in the environment: pattern and process. University of California Press, Berkeley, pp 24–54

Lawson ST (1999) Cloud water chemistry and mercury deposition in a high elevation spruce-fir forest. M.S. Thesis, University of Vermont

McLaughlin SB, Wimmer R (1999) Tansley Review No. 104 calcium physiology and terrestrial ecosystem processes. Ann Bot 83:373–417

Miller EK, VanArsdale A, Keeler JG, Chalmers A, Poissant L, Kamman N, Brulotte R (2005) Estimation and mapping of wet and dry mercury deposition across northeastern North America. Ecotoxicology 14:5–70

Morrissey CA, Bendell-Young LI, Elliott JE (2004) Linking contaminant profiles to the diet and breeding location of American dippers using stable isotopes. J Appl Ecol 41:502–512

Morrissey CA, Bendell-Young LI, Elliott JE (2005) Assessing trace-metal exposure to American dippers in mountain streams of southwestern British Columbia, Canada. Environ Toxicol Chem 24:836–845

Morrissey CA, Elliott JE, Ormerod SJ (2010) Diet shifts during egg laying: implications for measuring contaminants in bird eggs. Environ Poll 158:447–454

National Atmospheric Deposition Program (2007) NADP Program Office (NRSP-3), Illinois State Water Survey, 2204 Griffith Dr., Champaign, IL 61820. http://nadp.sws.uiuc.edu/MDN/maps.aspx. Accessed 15 Oct 2013

National Park Service Air Resources Division (2002) Air quality in the national parks, 2nd edn. Department of the Interior, Lakewood

Nodvin SC, Miegroet H, Lindberg SE, Nicholas NS, Johnson DW (1995) Acidic deposition, ecosystem processes, and nitrogen saturation in a high elevation Southern Appalachian watershed. Water Air Soil Pollut 85:1647–1652

Osborne CE, Evers DE, Duron M, Schoch N, Yates D, Buck D, Lane OP, Franklin J (2011) Mercury contamination within terrestrial ecosystems in New England and Mid-Atlantic states: profiles of soil, invertebrates, songbirds, and bats. BioDiversity Research Institute. BRI report 2011–09

Paritte JM, Kelly JF (2009) Effect of cleaning regime on stable-isotope ratios of feathers in Japanese quail (*Coturnix japonica*). Auk 126:165–174

Parker AJ (1982) The topographic relative moisture index: an approach to soil-moisture assessment in mountain terrain. Phys Geogr 3:160–168

Pearson SF, Levey DJ, Greenberg CH, Martínez del Rio C (2003) Effects of elemental composition on the incorporation of dietary nitrogen and carbon isotopic signatures in an omnivorous songbird. Oecologia 135:516–523

Poole A (eds) (2005) The birds of North America online; Cornell Laboratory of Ornithology, Ithaca, NY, http://bna.bords.cornell.edu/BNA/. Accessed 15 Oct 2013

Post DM (2002) Using stable isotope methods to estimate trophic position: models, methods, and assumptions. Ecology 83: 703–718

Post DM, Pace ML, Hairston NG Jr (2000) Ecosystem size determines food-chain length in lakes. Nature 405:1047–1049

Pyle P (1997) Identification guide to North American birds, Part 1. Slate Creek Press, Bolinas

R Core Team (2013) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/. Accessed 15 Oct 2013

Rimmer CC, McFarland KP, Evers DC, Miller EK, Aubry Y, Busby D, Taylor RJ (2005) Mercury concentrations in Bicknell's Thrush and other insectivorous passerines in montane forests of Northeastern North America. Ecotoxicology 14:223–240

Rimmer CC, Miller EK, McFarland KP, Taylor RJ, Faccio SD (2010) Mercury bioaccumulation and trophic transfer in the terrestrial food web of a montane forest. Ecotoxicology 19:697–709

Robinson SA, Lajeunesse MJ, Forbes MR (2012) Sex difference in mercury contamination of birds: testing multiple hypotheses with meta-analysis. Environ Sci Technol 46:7094–7101

Scheuhammer AM, Meyer MW, Sandheinrich MB, Murray MW (2009) Effects of environmental methylmercury on the health of wild birds, mammals, and fish. Ambio 36:12–19

Scheuhammer AM, Basu N, Evers DC, Heinz GH, Sandheinrich MB (2012) Ecotoxicology of mercury in fish and wildlife: recent advances. In: Bank MS (ed) Mercury in the Environment: Pattern and Process. University of California Press, Berkeley, pp 223–238

Seewagen CL (2010) Threats of environmental mercury to birds: knowledge gaps and priorities for future research. Bird Conserv 20:112–123

Seewagen CL (2013) Blood mercury levels and the stopover refueling performance of a long-distance migratory songbird. Can J Zool 91:41–45

Shortle WC, Smith KT (1988) Aluminum-induced calcium deficiency syndrome in declining red spruce. Science 240:1017–1018

Shriver WG, Evers DC, Hodgman TP, Macculloch BJ, Taylor RJ (2006) Mercury in sharp-tailed Sparrows breeding in coastal wetlands. Environ Bioindic 1:129–135

Townsend JM, Rimmer CC, Driscoll CT, McFarland KP, Iñigo-Elias (2013) Mercury concentrations in tropical resident and migrant songbirds on Hispaniola. Ecotoxicology 22:86–93

Valente RJ, Shea C, Humes KL, Tanner RL (2007) Atmospheric mercury in the Great Smoky Mountains compared to regional and global levels. Atmos Environ 41:1861–1873

Vander Zanden MJ, Rasmussen JB (1996) A trophic position model of pelagic food webs: impact on contaminant bioaccumulation in lake trout. Ecol Monogr 66:451–477

Vander Zanden MJ, Rasmussen JB (1999) Primary consumer δ^{13}C and δ^{15}N and the trophic position of aquatic consumers. Ecology 80:1395–1404

Vitz AC, Rodewald AD (2012) Using stable isotopes to investigate the dietary trophic level of fledgling songbirds. J Field Ornithol 83:73–84

Wada H, Cristol DA, McNabb FMA, Hopkins WA (2009) Suppressed adrenocortical responses and thyroid hormone levels in birds near a mercury-contaminated river. Environ Sci Technol 43:6031–6038

Warner SE, Shriver WG, Olsen BJ, Greenberg RG, Taylor RJ (2012) Mercury in wing and tail feathers of hatch-year and adult tidal marsh sparrows. Arch Environ Contam Toxicol 63:586–593

Weathers KC, Lovett GM, Likens GE (1995) Cloud deposition to a spruce forest edge. Atmos Environ 29:665–672

Weathers KC, Lovett GM, Likens GE, Lathrop R (2000) The effect of landscape features on deposition to Hunter Mountain, Catskill Mountains, New York. Ecol Appl 10:528–540

Weathers KC, Simkin SM, Lovett GM, Lindberg SE (2006) Empirical modeling of atmospheric deposition in mountainous landscapes. Ecol Appl 16:1590–1607

White RD, Patterson KD, Weaklery A, Ulrey CJ, Drake J (2003) Vegetation classification of Great Smoky Mountains National Park. Report submitted to BRD-NPS Vegetation Mapping Program, Nature Serve, Durham, NC

Xun L, Campbell NER, Rudd JWM (1987) Measurement of specific rates of net methylmercury production in the water column and surface sediments of acidified and circumneutral lakes. Can J Fish Aquat Sci 44:750–757

Yates DE, Mayack DT, Munney K, Evers DC, Major M, Kaur T, Taylor RJ (2005) Mercury levels in mink (*Mustela vison*) and river otter (*Lontra Canadensis*) from northeastern North America. Ecotoxicology 14:263–274

Zuur AF, Ieno EN, Meesters EHWG (2009) A beginner's guide to R. Springer, New York

Zuur AF, Ieno EN, Walker NF, Saveliev AA, Smith GM (2013) Mixed effects models with extensions in ecology with R. Springer, NY

Reproduced with permission of the copyright owner. Further reproduction prohibited without permission.