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Scale-dependence in polychlorinated biphenyl (PCB) exposure effects on waterbird habitat occupancy

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Abstract Spatial scale is rarely considered in population-level assessments of contaminant impacts on wild animals; as a result misinterpretation of the relationship between contaminant exposure and population status may occur. We assessed the strength of expression of polychlorinated biphenyl (PCB) exposure effects at local vs. regional spatial scales on population status in five species of waterbirds, "bioaccumulators" often promoted as indicators of contaminant effects in aquatic ecosystems. Our focus was the upper Hudson River where PCBs occur at levels reported to have adverse impacts on wild birds. At the local scale, waterbird habitat occupancy was estimated from 220 repeat surveys made between 2001 and 2010 along the same survey route divided into 25 contiguous river segments with markedly different PCB concentrations. At the regional scale, waterbird habitat occupancy in relation to proximity to the upper Hudson River was estimated across 1248 Breeding Bird Atlas survey blocks while controlling for region-wide variation in habitat availability. At the local scale, many associations of habitat and sampling covariates with species detection probabilities were evident but none, including PCB concentration, with habitat occupancy, extinction or colonization of a given river segment. At the regional scale, survey effort and habitat factors not related to PCB exposure were the most important drivers of waterbird occurrence although two species were more likely to occur farther from the contaminated river segment. Spatial scale clearly mediates expression of contaminant impacts on wild bird populations; large-scale, expert-generated databases provide an underused opportunity for better delineating the spatial scales at which population impacts occur and risk assessments should be performed.

Keywords PCBs · Habitat occupancy · Spatial scale · Hudson River · Waterbirds

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Introduction

Many species of birds are experiencing precipitous population declines across the globe as a result of multiple anthropogenic stressors (IUCN 2016). Environmental contaminants are a frequently identified cause of bird decline (e.g., Mineau and Whiteside 2013) with persistent forms of contaminants of particular concern (e.g., Best et al. 2010; Custer et al. 2003). It is important to disentangle how contaminant effects on bird mortality might affect, individually and cumulatively, bird populations (Walker et al. 2012; Köhler and Triebskorn 2013). Doing so requires field studies that assess local exposure and population impacts combined with large-scale data syntheses (Loss et al. 2015).

Most ecotoxicology risk assessments and population impact studies for birds are typically conducted at contaminated sites with individuals of a focal species assumed to reside in the area of contamination 100% of the time (Carlsen et al. 2004). However, species respond to habitats for particular life-history functions across a hierarchy of spatial scales (Sirami et al. 2008). Contaminants can affect avian reproduction at a given site, but how those effects are manifested for local vs. regional populations is rarely obvious. It is notable therefore how scarce are studies that have incorporated multiple spatial scales within the same analysis in population-level impact assessments of contaminant effects on wild birds (Kendall et al. 2010) although a few have done so (e.g., Elliott et al. 2007; Roscales et al. 2010; Yates et al. 2010). Multi-scale assessment is critical for avoiding misinterpretation of the nature or strength of the relationship between contaminant exposure and population status as well as for supporting delineation of the spatial scales at which wild bird population impact and risk assessments should be performed (Munns 2006).

PCBs are a frequently cited contaminant risk to birds (Best et al. 2010; Custer et al. 1998, 2003; Fredricks et al. 2011) and have been reported to harm birds in multiple ways (reviewed by Harris and Elliott 2011), for example, by decreasing growth (Gould et al. 1997), delaying egg laying and extending incubation periods (Murk et al. 1996), or reducing hatching success (Custer et al. 2003). The upper Hudson River in New York State, U.S.A. is a major focus for studies related to PCB-effects on biota (Limburg et al. 2012). This segment of the river historically received discharge of PCBs from industrial sources that resulted in contamination of river sediments, shorelines, and floodplains such that the upper Hudson River is now designated as a Superfund site by the U.S. Environmental Protection Agency (US EPA 2000). Concentrations of PCBs in tissues of a variety of bird species sampled on river are elevated (McCarty and Secord 1999; Hudson River Natural Resource Trustees 2004, 2011, 2013; Custer et al. 2010a, b).

Use of indicator species is a frequent practice in environmental assessment. For contaminated aquatic systems waterbirds are widely promoted as indicator species (e.g., Golden and Rattner 2003). Waterbirds occupy a strategic position as secondary consumers in aquatic ecosystems and hence as bioaccumulators of contaminants, including PCBs, by way of trophic transfer (Thomas and Anthony 1999; Henny et al. 2003; Elliott et al. 2012). This is due to waterbirds' increased likelihood of consuming contaminated food items, mainly fish, amphibians, and adult insects that develop from aquatic larvae, combined with the mobility and persistence of PCBs.

A major challenge for multi-scale assessment of change in population status in wild birds due to contaminant exposure is controlling for other environmental variables that simultaneously influence bird abundance and distribution (Carlsen et al. 2004) among other population endpoints. As is the case with many contaminated rivers, the upper Hudson River bisects a landscape comprised of array of aquatic and terrestrial habitats juxtaposed with multiple urban-to-rural gradients of land use (Vispo and Knab-Vispo 2011). Evaluating impact of contaminant releases must be conducted within an analytical framework that considers the existing state of the landscape and heterogeneity of critical habitats for a given species within it (Carlsen et al. 2004). This is necessarily a multi-scale problem because bird populations respond to many ecological factors operating at both local and regional spatial scales (Schaub et al. 2012).

This study's objective was to evaluate scale-dependency of impacts of PCB exposure on population status of waterbirds. Our study focus was five waterbird species frequently used as indicator species in studies of contaminated aquatic ecosystems: Great Blue Heron (*Ardea herodias*), Green Heron (*Butorides virescens*), Osprey (*Pandion haliaetus*), Spotted Sandpiper (*Actitis macularius*), and Belted Kingfisher (*Megaceryle alcyon*). We assessed habitat occupancy in each species in relation to the upper Hudson River where in the case of waterbirds PCB concentrations in, for example, belted kingfisher tissues are elevated by an order of magnitude in the contaminated section of the upper Hudson River (averaging 13,900 parts per billion in eggs) relative to off-river birds (averaging 2660 ppb; Hudson River Natural Resource Trustees 2004; see also Custer et al. 2010b) and at levels considered to pose "risk" to these organisms (Hudson River Natural Resource Trustees 2011, 2013).

We synthesized two complementary population occurrence surveys of these species: one made along contiguous segments of the upper Hudson River itself that varied strongly in PCB concentrations and another made at the landscape-scale on survey blocks that varied strongly in their proximity to the Hudson River and hence degree of PCB exposure for regional waterbird populations. At both spatial scales we focused on extent of occupied breeding habitat to characterize population-level response to PCBs because this state variable (one of several metrics that can be used to assess population-level effects of contaminants) has become the standard broadly promoted by the U.S. government to assess population status and trends in wildlife (Nichols et al. 2007). We made two, non-exclusive predictions to examine whether spatial might mediate expression of PCB impacts on wild bird populations: (1) if PCB contamination along the Hudson River was affecting populations primarily through local processes (mainly recruitment) then habitat occupancy would be reduced more in PCB-contaminated river segments compared to less contaminated segments, and (2) if impacts were expressed primarily through regional population processes (depressed recruitment and dispersal associated with contaminated areas) then habitat occupancy would be less in areas near the river compared to farther away.

Materials and methods

Among the large suite of waterbirds associated with the upper Hudson River (McGowan and Corwin 2008) we levied the following criteria to select a priori a suite of species for inclusion in this analysis: (1) primarily aquatic and mostly carnivorous (fish or insect-eating) and therefore potential bio-accumulators and hence susceptible to PCB-contamination impacts, (2) widely reported along the upper Hudson River and surrounding region and thus able to support a statistical analysis of environmental drivers of habitat occupancy, (3) breeding range extended throughout the study region so as not to conflate range discontinuities with patterns of habitat occupancy, and (4) broadly representative of avian taxonomic diversity, that is, drawn from multiple avian families. Levying these criteria identified five species—Great Blue Heron, Green Heron, Osprey, Spotted Sandpiper, Belted Kingfisher—all of which have also been frequent subjects of prior published field assessments of susceptibility to environmental contaminants although not necessarily along the Hudson River (e.g., Great Blue Heron: Hart et al. 1991; Custer et al. 1997; Elliott et al. 1989; Green Heron: Niethammer et al. 1984; Wainwright et al. 2001; Hothem et al. 2006; Osprey: Steidl et al. 1991; Henny et al. 2010; Elliott et al. 2012; Spotted Sandpiper: Hesse et al. 1975; Custer et al. 2010a; Belted Kingfisher: Baron et al. 1997; Custer et al. 2010b). Focal species included one (Spotted Sandpiper) characterized by AHRI expression constructs associated with intermediate sensitivity to effects of dioxin-like compounds with the remainder (or species closely related to them) of predicted lower sensitivity (Farmahin et al. 2013).

To measure waterbird habitat occupancy in relation to PCB exposure at the local scale, observations of waterbirds were gathered on the upper Hudson River by a single individual piloting a pontoon boat on 220 occasions from 2001 to 2010 along a fixed, 50-km-long route between 43.26° N (Rogers Island) and 42.87° N (Lock 2, Mechanicville). The time, date and position along the main axis of the riverbed perpendicular to every individual waterbird sighted was recorded with a hand-held GPS unit. Survey segments associated with narrow canals with low visibility and those by-passed by lock and dam systems for passage around rapids were excluded. Analysis was restricted to observations made during the breeding season, that is, May through September. All observations from 2009 were excluded to avoid conflating any impacts of dredging activities associated with large-scale remediation efforts conducted in the surveyed area that year. Weather and river flow data associated with each observation were derived from National Climatic Data Center (http://www.ncdc.noaa.gov/cdo-web) and United States Geological Survey (for pre-2007: http://water.usgs.gov/data, and for post-2007: http://nwis.waterdata.usgs.gov/nwis/dv/?referred_module=sw), respectively. Spatial correlation (variogram) analysis of waterbird abundance was conducted to evaluate the spatial scale at which waterbird counts were independent and hence appropriateness of various sampling segmentations of the river for statistical analysis. To do so, counts of the five target species were assigned to their nearest 1/10th mile (0.16 km) river markers and the average count of observed species per year was computed for each river marker. Species-specific variograms along the river for each year calculated using the “automap” package in R (Hiemstra et al. 2009).

Because waterbird abundance data were dominated by zero counts (12% or 2402 of 20,556 abundance observations were >1) we treated waterbird counts as binary (presence or absence) and analyzed them using site-occupancy models that allowed differentiation between the probabilities of site occupancy (ψ), colonization (γ), extinction (ε) and species detection (p) by incorporating site-specific covariates (to estimate ψ, γ, ε and p) and sampling co-variates (to estimate p only, MacKenzie et al. 2006). Sampling co-variates included weather-related variables during a given survey (river flow, fog, precipitation, temperature, wind, and month). Habitat-related variables were measured for 25 equal-length river segments and included extent of floating aquatic vegetation, forest, fringe wetland, natural shoreline, river depth, river width, river segment, whether a river segment hosted submerged aquatic vegetation or SAV: 0–25, 25–50, 50–75, or 75–100%, and total PCB (Σ PCB) sediment concentration (EPA 2016).

Occupancy modeling was performed using the Simple Multi-Season Model in PRESENCE version 9.5 (MacKenzie 2012) to link species detections on a given river segment with sampling and habitat variables for that same segment across the 220 repeat surveys. Prior to the analysis, all continuous scalar covariates were converted to their equivalent standard scores, i.e., the signed number of standard deviations above or below the mean. Categorical variables were replaced by their equivalent indicator variables, i.e., 1 when the category matched, otherwise 0. For each species we assessed the same two, fully parametrized candidate models (that is, all sampling and site-co-variates included) that varied only in including or not including Σ PCB concentration in order to directly address the study’s a priori hypothesis, that is, site occupancy (ψ), colonization (γ) and extinction (ε) by a given species was influenced by potential Σ PCB-exposure. Akaike Information Criterion (AIC) of each model was computed and ranked; models with ΔAIC (the difference between a particular model AIC and the minimum AIC) < 2 were considered equivalent (Burnham and Anderson 2002).

To measure waterbird habitat occupancy in relation to Σ PCB exposure at the regional scale, spatial trends in occurrence of each of the five focal species relative to proximity to the upper Hudson River were examined using New York State Breeding Bird Atlas (BBA) data, a volunteer-based, state-wide survey collecting information on the presence and breeding status for all breeding birds found within sampling blocks (Andrle and Carroll 1988; McGowan and Corwin 2008). The BBA sampling grid was scaled at 10×10 km and subdivided for fieldwork into four, 5×5 km blocks. Observations were made by skilled birders spending >8 h in each block with observer effort recorded for each block as number of person hours of searching. Avian breeding was reported at three levels of certainty based on the behavior of birds observed (possible, probable, and confirmed; McGowan and Corwin 2008) any of which we considered to be evidence of occurrence thereby generating a presence/absence dataset.

BBA data offer: (1) a very large sample of landscape-level sampling units, (2) non-overlapping landscapes (i.e., Atlas blocks), and (3) opportunity to link landscape attributes to waterbird occurrence. Our analysis focused on waterbird occurrence as reported in the BBA 2000–2005 (the second iteration of the Atlas) when for each BBA block a comprehensive suite of contemporaneously measured landscape variables was available. More specifically, we linked block-specific Atlas occurrence reports for each species to a companion suite of largely uncorrelated (all pairwise correlations of Pearson $r < 0.50$), waterbird-relevant, landscape-scale environmental factors: extent of forest, developed and barren land, open water, emergent herbaceous wetland, scrub–shrub wetland, and forested wetland (National Land Cover Dataset: Homer et al. 2007, resolution 30 m, downloaded from www.mrlc.gov), and lengths of pond shores, lakeshores, riverbanks and streams. Shore length for each was derived from New York State Area Hydrography dataset [1:24,000 scale] from the New York State Office of Cyber Security & Critical Infrastructure Coordination, downloaded from New York State GIS Clearinghouse. Length of rivers derived from New York State Hydrography Digital Line Graph dataset [1:2,000,000 scale], downloaded from CUGIR, Cornell University, NY. Proximity to the upper Hudson River was indexed by distance of the centroid of a given BBA block to the closest segment of the upper Hudson River. For the purposes of this study we defined the contaminated component of the upper Hudson River as occurring between Glens Falls and Stillwater within which levels of Σ PCB contamination have been documented as highest (US EPA 2000). Distance to the upper Hudson River was not correlated (Pearson r, $P > 0.05$) with any other environmental variable measured on Atlas blocks; therefore, proximity of survey block centroid to the upper Hudson River provided an independent index of Σ PCB exposure risk. Moreover, there is no other known, significant source of Σ PCB contamination in the region.

Our analysis was restricted to BBA blocks situated within 100 km of the upper Hudson River. This distance was ×3 the typical maximum foraging range of the most widely-ranging species in the analysis—Great Blue Herons (Butler 1992)—and hence presumably including population segments for all species along a gradient of high to low exposure to the upper Hudson River and associated Σ PCB contamination. All continuous landscape variables were standardized to zero mean and unit standard deviation and then linked to occurrence of each waterbird species on a given BBA sampling block via a generalized linear model (glm, R 2.13.1: R Development Core Team 2011) with a binomial error term owing to the dichotomous nature of the avian response variable (evidence of breeding or not). Standardizing variables enabled us to contrast their relative contributions (effect sizes) directly based on the relative magnitude of the model coefficients. Akaike information criterion (AIC, Burnham and Anderson 2002) was used to select among candidate models with or without inclusion of proximity to the upper Hudson River. A single sampling variable—the natural log of effort, or total hours devoted by Atlas observers to surveying a particular block—was included in all models to control for this potentially important driver of species detection during bird Atlas efforts.

Results

Waterbird surveys repeated 220 times along the same survey route divided into 25 river contiguous segments between Roger Island and Mechanicville on the upper Hudson River during 2001–2010 (excluding 2009 during river dredging) detected 1894 Great Blue Herons at 1290 sites, 740 Belted Kingfishers at 570 sites, 401 Spotted Sandpipers at 239 sites, 199 Green Herons at 123 sites, and 65 Ospreys at 59 sites. Spatial correlation (variogram) analysis conducted for all five species at all spatial scales in all years to estimate the spatial scale at which survey data varied independently generated correlated variograms for only 2 of 45 possible species × year combinations. In both cases the variogram range was substantially less than the 1 mile (1.6 km) sampling unit used, i.e., 0.38 miles for osprey in 2006 and 0.28 for green heron in 2008; therefore, or use of 1-mile (1.6 km) river segments for analyzing bird occurrence represented independent samples. Many sampling and site co-variates measured on segments of the upper Hudson River contributed to variation in detection probabilities for each of the five waterbird species (Tables 1, 2) but none contributed to variation in site occupancy. Notably, candidate models explaining habitat occupancy that included extent of Σ PCB contamination of a given river segment performed no better than those that did not for each of the five waterbird species assessed (Table 1).

Table 1 Summary statistics for habitat occupancy and species detection models based on waterbird surveys, repeated 220 times for five species along 25 river segments between Rogers Island and Mechanicville, New York, on the upper Hudson River during 2001–2010 (excluding 2009 during remediation dredging). Sampling co-variates included: fog, precipitation, river flow, average temperature, wind, and month. Habitat-related variables included extent of floating aquatic vegetation, shoreline length, forest, fringe wetland, river depth, river width, river segment, whether a segment hosted submerged aquatic vegetation or SAV, and average Σ PCB concentrations. For each species two, fully parametrized candidate occupancy models (that is, all sampling and site-co-variates included) that varied only in including or excluding Σ PCB concentration were assessed in order to directly address the study's a priori hypothesis, that is, site occupancy (ψ) was influenced by potential Σ PCB-exposure

Species	Σ PCB covariate	AIC	ΔAIC	N	−2LogLike	Sampling occasions
Great Blue Heron	Excluded	4347.54		68	4211.54	220
	Included	4407.46	59.93	71	4265.46	220
Belted Kingfisher	Excluded	3421.15		68	3285.15	220
	Included	3508.70	87.55	71	3366.70	220
Spotted Sandpiper	Excluded	2283.95		68	2147.95	220
	Included	2760.97	477.02	71	2618.97	220
Green Heron	Excluded	1348.51		68	1212.51	220
	Included	1458.85	110.35	71	1316.85	220
Osprey	Excluded	945.21		68	809.21	220
	Included	946.02	0.82	71	804.02	220

A total of 1248 Atlas blocks occurred within 100 km of the upper Hudson River and were included in the analysis. Breeding season occurrence of waterbirds in a given Atlas block during 2000–2005 was associated with combinations of environmental variables unique to each species that nevertheless overlapped broadly (Table 3). More specifically, breeding season occurrence of Great Blue Herons (observed on 68.1% of blocks) and Green Herons (27.2 % blocks) was more likely on survey blocks with more extensive pond and lake shores and less likely at higher elevations and in more forested blocks. Additionally Green Heron occurrence was less likely where there was more development and barren lands and more open water (Table 3). Osprey occurrence (15.1% of blocks) was more likely where there were more extensive pond and lake shores, more extensive forested wetlands and on blocks farther from the upper Hudson River (increasing from 7 to 16% occurrence at 0 and 100 km, respectively, when other variables held constant at their average values, Fig. 1). Osprey occurrence, however, was less likely where stream networks were more extensive. Spotted Sandpipers (30.0% of blocks) were more likely to occur where there was more extensive pond and lake shorelines, more streams, more extensive river banks, and also on BBA blocks farther from the upper Hudson River (increasing from 22 to 33% at 0 and 100 km, respectively, Fig. 1) whereas Spotted Sandpipers were less likely to occur at higher elevations and where forests were more extensive. Belted Kingfishers (66.5% of blocks) were more likely to occur where river margins were more extensive and less likely to occur where forests were more extensive. Probability of occurrence of all species was higher on blocks where survey effort was greater (Table 1). Models with the full suite of environmental variables performed no better (differed by <2 ΔAIC units) than the same models lacking only proximity to the upper Hudson River variable for all species except Osprey and Spotted Sandpiper for which inclusion of river proximity improved model fit (models with river proximity included differed by >2 ΔAIC units than those lacking this variable, Table 3).

Discussion

Identification of the spatial scale at which populations operate and are most strongly influenced by limiting factors, including contaminants, is always a challenge for population studies of wild birds (Baillie et al. 2000). We conducted intensive studies of habitat occupancy in five waterbird species for nearly a decade along an extended segment of the upper Hudson River that included strong heterogeneity in Σ PCB contamination. We explicitly accounted for variation in detection probability which was influenced by many sampling- and habitat-related variables. Yet we failed to identify any consistent drivers of habitat occupancy at the local scale for any of the five species assessed, including degree of Σ PCB contamination of a given river segment.

Whereas the habitat variables measured in this study at the local-scale could be detected by the focal bird species and are known to be used by them to guide habitat selection, the presence of PCBs in focal bird species' food, or at differing concentrations in river sediments, cannot be detected by birds; therefore, PCB exposure is not a variable that is as likely to affect habitat selection on a daily basis but rather would manifest its effect, if any, via population processes,

Table 2 Occupancy (ψ) and detection (P) probabilities for habitat occupancy and species detection models based on waterbird surveys repeated 220 times for five species along 25 river segments between Rogers Island and Mechanicville, New York, on the upper Hudson River during 2001–2010 (excluding 2009 during remediation dredging) in relation to sampling and habitat co-variates based on minimum-AIC models (see Table 3 for variable descriptions). Estimates > |0 .2| are highlighted in bold

	Great Blue Heron		Green Heron		Osprey		Spotted Sandpiper		Belted Kingfisher	
Variable	P	ψ	P	ψ	P	Ψ	P	ψ	P	ψ
FAV	−0.142	0.054	**−0.249**	−0.074	−0.001	0.009	**−0.646**	0.09	**−0.278**	0.04
Fog	−0.184	–	**−1.348**	–	−0.899	–	**−1.906**	–	**−0.581**	–
Forest	**0.207**	−0.031	**0.293**	−0.073	**0.231**	0.094	**0.296**	−0.06	−0.112	−0.014
Fringe Wetland	−0.131	−0.067	0.087	0.005	0.069	0.052	**−0.281**	−0.043	**−0.291**	0.19
Month	**0.438**	–	0.118	–	0.166	–	**−0.527**	–	−0.018	–
Precipitation	0.01	–	0.029	–	0.093	–	0.151	–	0.108	–
River depth	**−0.316**	0.063	**−0.6**	0.092	**−0.326**	−0.036	**−0.958**	0.123	**−1.032**	0.185
River flow	**−0.476**	–	−0.601	–	−0.483	–	−1.096	–	−0.55	–
River width	0.156	−0.093	**0.418**	−0.028	**0.218**	−0.066	**0.523**	−0.111	**0.714**	−0.132
River segment	**−0.252**	0.032	**−0.357**	0.096	−0.193	−0.041	**−0.543**	0.086	0.157	0.068
0–25% SAV	0.065	0.028	**0.235**	−0.042	−0.188	0.091	−0.002	−0.03	0.115	−0.157
25–50% SAV	0.023	−0.017	0.107	−0.032	0.185	−0.035	**0.544**	0	0.001	0.043
50–75% SAV	0.054	−0.037	0.088	0.015	**0.256**	−0.099	**0.204**	−0.016	−0.021	0.152
75–100% SAV	−0.189	0	−0.063	0.044	−0.137	0.053	**−0.222**	0.077	−0.08	−0.066
Unclassified SAV	**0.202**	0.022	**0.313**	−0.086	0.166	−0.07	0.122	0.092	−0.03	0.15
Natural shoreline	0.145	0.005	−0.115	−0.012	−0.331	0.04	−0.075	0.163	**−0.354**	0.047
Temperature (average)	0.173	–	0.044	–	0.013	–	0.197	–	−0.001	–
Wind	−0.097	–	−0.088	–	−0.066	–	0.034	–	−0.038	–

particularly recruitment. Many bird populations are synchronized across large spatial scales (e.g. Paradis et al. 2000; Jones et al. 2007; Saether et al. 2007), implying that population regulation often operates well beyond the local scale. Spatial autocorrelation of environmental processes (climate, land disturbance, disease, primary productivity) as well as population process, particularly dispersal, are likely the main reasons for population synchrony and lack of linkages observed in this and many other avian population studies between dynamics of local populations and local-scale habitat factors (Saether et al. 2007; Börger and Nudds 2014).

Our study did reveal possible interaction between Σ PCB exposure and waterbird occurrence at the regional spatial scale. In this analysis two species—Osprey and Spotted Sandpiper—exhibited a significant positive relationship at the regional scale between extent of occurrence and proximity to upper Hudson River (that is, reduced occurrence at shorter distances from the river) when variation in other key habitat variables was controlled for. In terms of effect sizes, for Osprey, proximity to the upper Hudson River was the least important driver among five significant variables affecting regional distribution and, for Spotted Sandpipers, it was sixth least important among seven significant habitat variables. These outcomes emphasize the need to control for multiple environmental factors in assessments of contaminant impacts on wild bird populations, as well as, to appropriately contextualize species-habitat-contaminant relationships in terms of both statistical significance and effect sizes (Cox 2010).

Increasing survey effort typically generates a higher number of recorded species in bird survey efforts (Tobler et al. 2008). In our study, survey effort was a significant driver for all species evaluated and also the first or second most important driver as indexed by coefficient strengths based on standardized variables. Modeling species occurrences based on bird atlas data opens up new opportunities for a more nuanced understanding of species responses to contaminant exposure while controlling for landscape configurations (Devictor et al. 2010); however, reliable inference clearly requires accounting for the sampling effort, which is not often done in analyses of bird atlas data of any kind (Sadoti et al. 2013). We believe our study is the first to leverage the opportunity for insight provided by large-scale bird atlas databases for examining contaminant effects on birds.

Local-scale assessments characterize most field studies of contaminant effects on birds, including on the Hudson

Table 3 Generalized linear models relating breeding season occurrence of five waterbird species on 1248 Breeding Bird Atlas blocks during 2000–2005 and within 100 km of the upper Hudson River in New York State to landscape configuration and sampling variables associated with each 5 × 5 km block (see text) all of which were standardized prior to analysis to enable comparison of model coefficients as indices of effects sizes: (A.) Model outcomes presented as coefficient ± standard error (rank among absolute values of significant parameters excluding intercept and (*) denotes $p < 0.05$, (**) denotes $p < 0.01$ and (***) denotes $p < 0.001$) and (B.): model fitting criteria for each species when distance of a BBA block from the upper Hudson River was included or excluded from the model (models with ΔAIC < 2.0 are considered equivalent)

	Great blue Heron	Green Heron	Osprey	Spotted Sandpiper	Belted Kingfisher
A. Model coefficients					
Intercept	0.851 + 0.066***	−1.384 + 0.088***	−2.061 + 0.102***	−0.957 + 0.068***	0.774 + 0.065***
Elevation	−0.178 + 0.08 (4)*	−0.477 + 0.113(5)***	−0.033 + 0.101	−0.285 + 0.086 (3)***	−0.05 + 0.078
Forest	−0.417 + 0.109 (2)***	−1.38 + 0.122 (1)***	0.27 + 0.169	−0.293 + 0.103 (2)**	−0.361 + 0.108 (2)***
Developed and barren land	−0.234 + 0.084	−0.201 + 0.088 (7)*	0.045 + 0.135	0.016 + 0.08	0.1 + 0.108
Open water	−0.078 + 0.102	−0.59 + 0.111 (2)***	0.101 + 0.103	0.005 + 0.086	−0.061 + 0.092
Emergent herbaceous wetland	0.019 + 0.082	0.119 + 0.08	0.054 + 0.078	0.084 + 0.069	0.085 + 0.088
Scrub–shrub wetland	0.014 + 0.077	−0.131 + 0.082	−0.09 + 0.121	−0.035 + 0.074	0.056 + 0.078
Forested wetland	0.024 + 0.078	−0.511 + 0.088 (3)***	0.338 + 0.102 (4)***	−0.054 + 0.074	0.029 + 0.078
Length of pond and lake shores	0.28 + 0.094 (3)**	0.312 + 0.1 (6)**	0.517 + 0.094 (1)***	0.194 + 0.082 (5)*	0.126 + 0.088
Length of streams	−0.095 + 0.071	−0.036 + 0.08	−0.359 + 0.11(3)***	0.16 + 0.072 (7)*	−0.096 + 0.07
Length of rivers	−0.043 + 0.067	−0.025 + 0.079	0.019 + 0.088	0.277 + 0.067 (4)***	0.267 + 0.07 (3)***
Effort	0.473 + 0.07 (1)***	0.49 + 0.082 (2)***	0.36 + 0.089 (2)***	0.396 + 0.07 (1)***	0.416 + 0.068 (1)***
Distance to Upper Hudson River	0.076 + 0.075	−0.102 + 0.087	0.31 + 0.099 (5)**	0.188 + 0.075(6)*	0.031 + 0.074
B. Model fitting criteria for models with distance to Upper Hudson River:	AIC				
Included	1473.4	1905.0	921.5	1413.0	1494.5
Excluded	1472.4	1094.4	929.4	1417.4	1492.6
ΔAIC	1.0	0.6	7.9	4.4	1.9

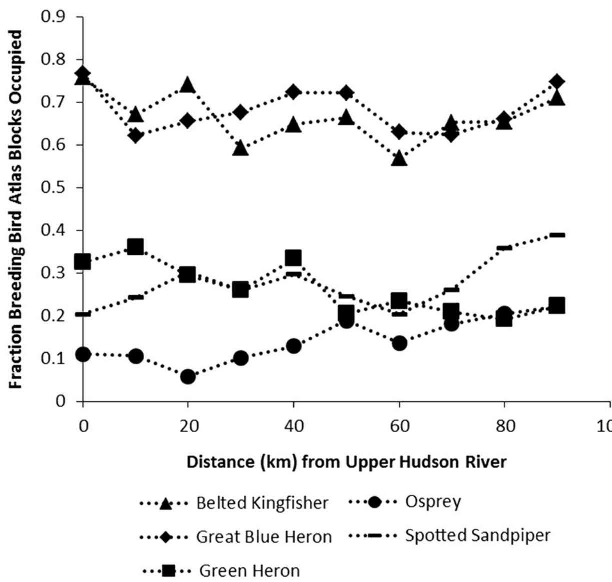

Fig. 1 Relationship between fraction of Breeding Bird Atlas survey blocks within 100 km of the upper Hudson River in New York State occupied by five waterbird species during the breeding season (2000–2005) vs. distance of a survey block from the upper Hudson River

River (US EPA 2000). Our study indicates that rather than evaluating the ecological impact on species in the immediate vicinity of a contaminant release, it may be more relevant to determine if population impacts occur at the landscape scale. Consideration of population impacts of contaminants at all relevant spatial scales is specifically called out in U.S. EPA guidance documents (US EPA 1998, 1999). Because the Hudson River has been contaminated mainly by Aroclor 1242 (TAMS 1997), which has lower proportions on toxic congeners than other PCB formulations, some caution should be exercised when contrasting outcomes of this study to others of wild bird populations exposed to different PCB mixtures. This said, in addition to designing contaminants research at relevant spatial scales, our study highlights the need for studies of contaminant impacts on wild bird species to choose focal species carefully, to articulate potential impacts both in terms of statistical significance and meaningful effect sizes, and to leverage opportunity for insight provided by large-scale, expert-generated databases.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no competing interests.

Ethical approval All applicable international, national, and/or institutional guidelines for the care and use of animals were followed.

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