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Deriving Field-Based Ecological Risks for Bird Species

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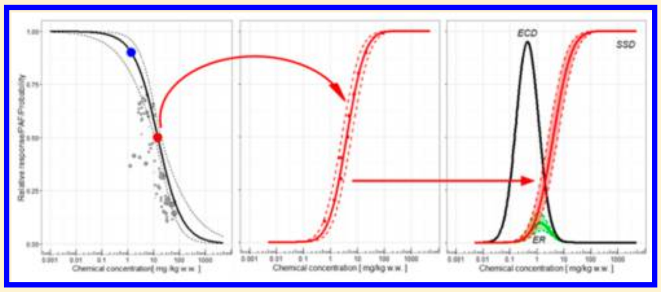
Supporting Information



ABSTRACT: Ecological risks (ERs) of pollutants are typically assessed using species sensitivity distributions (SSDs), based on effect concentrations obtained from bioassays with unknown representativeness for field conditions. Alternatively, monitoring data relating breeding success in bird populations to egg concentrations may be used. In this study, we developed a procedure to derive SSDs for birds based on field data of egg concentrations and reproductive success. As an example, we derived field-based SSDs for *p,p′*-DDE and polychlorinated biphenyls (PCBs) exposure to birds. These SSDs were used to calculate ERs for these two chemicals in the American Great Lakes and the Arctic. First, we obtained field data of *p,p′*-DDE and PCBs egg concentrations and reproductive success from the literature. Second, these field data were used to fit exposure-response curves along the upper boundary (right margin) of the response's distribution (95th quantile), also called quantile regression analysis. The upper boundary is used to account for heterogeneity in reproductive success induced by other external factors. Third, the species-specific $EC_{10/50}$s obtained from the field-based exposure-response curves were used to derive SSDs per chemical. Finally, the SSDs were combined with specific exposure data for both compounds in the two areas to calculate the ER. We found that the ERs of combined exposure to these two chemicals were a factor of 5−35 higher in the Great Lakes compared to Arctic regions. Uncertainty in the species-specific exposure-response curves and related SSDs was mainly caused by the limited number of field exposure-response data for bird species. With sufficient monitoring data, our method can be used to quantify field-based ecological risks for other chemicals, species groups, and regions of interest.

INTRODUCTION

In current ecological risk assessment, chemical risks are often evaluated using species sensitivity distributions (SSDs).[1−3] The main assumption of an SSD is that the differences in sensitivity of species toward a chemical can be described by a (cumulative or probabilistic) distribution function.[4,5] SSDs can be used to calculate the ecological risk (ER) posed by individual compounds and compound groups to species in an area.[6] ERs are defined as the probability of measured environmental concentrations exceeding (no) effect concentrations of species.[6,7] Effect concentrations, such as half maximal effective concentrations (EC_{50}s) or no-observed-effect concentrations (NOECs), are typically obtained from lab experiments. In these experiments, individual performance (e.g., fecundity or mortality) is measured in a controlled setting.[8] The application of these data in the derivation of SSDs has been subject to much discussion over the years as lab data fail to acknowledge potential differences with site-specific conditions.[9] Furthermore, although the use of a wide range of species when quantifying toxicant-effect relationships is more representative of actual impacts in bird populations,[10] test species used in these experiments are limited to those that are easy to breed and do not necessarily reflect natural compositions of taxonomical groups.[9,11,12] Ecotoxicological data is lacking for many bird species at higher trophic levels (e.g., piscivores, raptors, and insectivores), as laboratory experimentation is limited due to practical, financial, and ethical constraints[13] or focus on acute lethal toxicity only.[7]

Field monitoring data of breeding success in bird populations and related chemical concentrations in eggs may be used as an alternative to laboratory experiments in the derivation of effect concentrations as input for the SSDs for birds. However, it is important to isolate the impact of individual chemicals as breeding success is influenced by a variety of additional environmental or biological factors, such as climate and population density.[14] Traditional statistical regression approaches focus on changes in the mean of the response variable's distribution only, including the effects of other extraneous variables and introducing potential bias. Instead of using traditional regression analysis, quantile regression may be used to account for hidden bias resulting from extraneous environmental variables.[6,15,16]

By fitting a sigmoidal exposure-response curve at the upper boundary of the data's distribution, the constraints of elevated toxicant concentrations in eggs imposed on bird reproductive success are expected to become visible, as it corrects for unmeasured ecological and environmental factors (hidden bias)

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potentially limiting the observed response.[16] In this way, constraints imposed solely by chemical exposure are expected to be revealed.

The aim of the present study was to develop and apply a procedure to derive ecological risks of bird species, based on field data of reproductive success and chemical concentrations in eggs of birds. As an example, we applied the procedure with field data on *p,p′*-DDE (dichlorodiphenyldichloroethylene, a DDT metabolite) and the sum of multiple PCB congeners in the North American Great Lakes and the Arctic. The effects of DDT metabolites and polychlorinated biphenyls (PCBs) on bird species have received much attention since the early 1960s.[17,18] These compounds are known to be persistent and are strongly linked to eggshell thinning, altered sexual behavior, and hormonal disruption,[19,20] which in turn decrease growth rates in bird populations.

■ MATERIALS AND METHODS

Quantile Regression. *Fitting Procedure.* Relative reproductive success (fraction) was related to a toxicant gradient using the quantile regression approach as described by Koenker,[21] where the regression line was fitted to the 95th quantile (τ = 0.95) of the response variable's distribution, to account for heterogeneity caused by other limiting ecological and environmental factors. The quantile regression model was based on the Hill equation,[22−24] yielding a sigmoidal curve

$$R_{i,j} = \frac{1}{1 + \left[\frac{EC_{50i,j}}{C_j}\right]^{\beta_{i,j}}} \tag{1}$$

where $R_{i,j}$ is the modeled response (relative reproductive success) of species i for chemical j, $EC_{50i,j}$ is the 50% effective response level or inflection point of the curve of species i for chemical j, C_j is the measured contaminant concentration of chemical j (in mg/kg egg wet weight), and $\beta_{i,j}$ is the slope coefficient associated with species i for chemical j, determining the slope of the curve.[22,24] We fitted a logit transformed regression line on the data set using the *quantreg* package of Koenker (2013),[21] adapted from Bottai et al. 2010[25] as

$$\text{logit}(R) = \beta_0 + \beta_{i,j}\cdot \text{LN}(C) \tag{2}$$

implying that the untransformed relative reproductive success can be calculated through

$$R = \frac{\exp(\beta_0 + \beta_{i,j}\cdot \text{LN}(C))\cdot(R_{\max} + \varepsilon) + (R_{\min} - \varepsilon)}{\exp(\beta_0 + \beta_{i,j}\cdot \text{LN}(C)) + 1} \tag{3}$$

where LN(C) is the natural log-transformed chemical concentration in the egg, β_0 is defined as the intercept of the regression line, and $\beta_{i,j}$ is the slope coefficient equal to $\beta_{i,j}$ in eq 1. ε reflects the error term, in this study set at 0.001, a small quantity ensuring that the logistic transform is defined for all values of R.[25] The quantile regression algorithm minimizes the residuals of the regression analysis.[21] The inflection point of the exposure-response curve (or EC_{50} in eq 1, in mg/kg w.w.) was derived as

$$EC_{50} = \exp\left(\frac{\beta_0}{\beta_{i,j}}\right) \tag{4}$$

The EC_{10} values were derived by solving eq 1 for C, fixing $R_{i,j}$ at 0.9,[26] using the *uniroot* function in R statistics 3.3.1. The 95% confidence intervals associated with the quantile fit were obtained by using bootstrapped errors, as described by Koenker.[27]

Model Consistency. Regression lines were fitted at three additional quantiles (τ = 0.25, 0.5, 0.75) to evaluate model consistency. We expected a negative relationship between exposure and response for all quantiles, i.e. a negative $\beta_{i,j}$. Furthermore, we expected that the larger the quantile, the larger the EC_{50}- and EC_{10}-values. If both expectations were not met, data were not considered sufficiently reliable to derive an exposure-response relationship. The following selection criteria for the exposure-response curves were applied:

1) Exposure-response curves yielding a positive $\beta_{i,j}$ for τ = 0.95, τ = 0.75, or τ = 0.50 were not further considered in the analysis.

2) Effect concentrations that have higher EC_{10}- or EC_{50}-values for τ = 0.25 or τ = 0.50 compared to τ = 0.95 were disregarded.

Species Sensitivity Distributions. SSDs were constructed, using EC_{50} and EC_{10} data obtained from the derived exposure-response curves. We assumed a log-normal spread in species sensitivities for SSDs with mean (μ) and standard deviation (σ) to link the toxicant gradient (x-axis) to the potentially affected fraction of species (PAF; y-axis).[4,28,29] To assess the statistical uncertainty in the SSDs, we randomly sampled 10,000 EC_{10}s and EC_{50}s for each species-chemical combination separately, using the uncertainty in the quantile regressions as a starting point. Subsequently, for each chemical 10,000 SSDs were fitted through these sampled effect concentrations over all species under the assumption of a log-normal distribution.

Ecological Risks. The ecological risk (ER; fraction) is defined as the probability of a species in a certain area exceeding its EC_{10} or EC_{50} and represents the overlap between the derived SSD and the exposure concentration distribution (ECD) of a certain chemical in a specific area (as exemplified in Figure 1). ERs are calculated through integral[6,7]

$$ER = \int_{-\infty}^{\infty} PDF_{ECD}\cdot CDF_{SSD}\, dx \tag{5}$$

where PDF_{ECD} is defined as the probability density function of the natural log-transformed exposure concentration distribution of an individual chemical found in bird's eggs in a specific area. CDF_{SSD} is the (cumulative) single substance SSD model based on field-based EC_{10}s and EC_{50}s.

ERs based on SSDs derived for respectively EC_{10}s and EC_{50}s for single toxicant exposure were calculated based on Korsman et al. 2016[7] in R statistics 3.3.1 using

$$ER = \text{pnorm}\left(\frac{\mu_{ECD} - \mu_{SSD}}{\sqrt{{\sigma_{ECD}}^2 + {\sigma_{SSD}}^2}}\right) \tag{6}$$

in which the function pnorm returns a probability distribution function and where μ_{SSD} is the natural log-transformed mean of the SSD and σ_{SSD} reflects the natural log-transformed standard deviation of the SSD. The μ_{ECD} and σ_{ECD} are the mean and standard deviation associated with the natural log-transformed contaminant concentrations found in bird's eggs in a certain area.[6]

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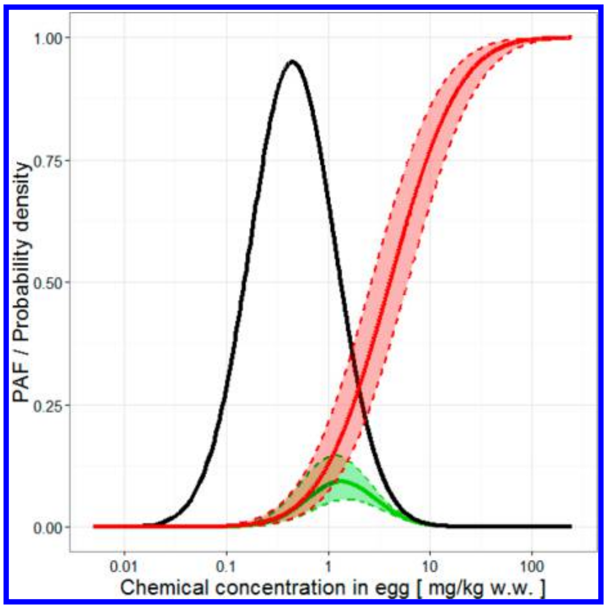

Figure 1. Graphical example of CDF_{SSD} in red, corresponding 95% confidence intervals (dashed lines), PDF_{ECD} in black, and ER (area under the curve), with corresponding 95% confidence intervals, in green.

Combined ecological risk (ER_c), defined as the risk posed by multiple pollutants in an area, was calculated using eq 7 according to the response addition principle[6,7]

$$ER_c = 1 - \prod_{i=1}^{n_i} (1 - ER_i) \quad (7)$$

where n_i is the number of substances used in calculation of the ER_c, and ER_i is the ecological risk calculated for each compound individually. Statistical uncertainty in the ERs was quantified by random sampling from the SSD realizations, as mentioned above.

Data Acquisition and Treatment. *Exposure-Response Data.* A data set was compiled containing *p,p′*-DDE and ΣPCB concentrations in eggs (in mg/kg wet weight) and the corresponding reproductive success of the bird (-population). The relative reproductive success of an individual bird (or population) was defined as the fraction of fledglings per occupied or active nest (productivity), over the species theoretical maximum productivity (the maximum productivity given in the full data set for that species, see Table S1 and eq S1 in the Supporting Information). Data were obtained from a literature search of the Web of Knowledge using search strings related to reproductive success and productivity, combined with species' names (scientific and common names) or terms such as *birds* and *avian species* on one side, and terms related to organic pollutants (specific compound names or compound groups) on the other. Additionally, data from scientific reports (gray literature) were obtained using Google Scholar, using the same search strings. This search revealed 57 potentially useful articles and reports,[30−86] including monitoring data for four raptorial bird species or species groups (bald eagle (*Haliaeetus leucocephalus*), white-tailed eagle (*Haliaeetus albicilla*), Eurasian sparrowhawk (*Accipiter nisus*), and falcon species (including *Falco peregrinus* and *Falco sparverius*)), eight piscivorous bird species or species groups (osprey (*Pandion haliaetus*), herring gull (*Larus argentatus*), common tern (*Sterna hirundo*), cormorant species (including *Phalacrocorax auritus*, *Phalocrocorax carbo*, and *Phalacrocorax pelagicus*), brown pelican (*Pelecanus occidentalis*), snowy egret (*Egretta thula*), black-crowned night heron (*Nycticorax nycticorax*), and black skimmer (*Rynchops niger*)), and three insectivorous birds (tree swallow (*Tachycineta bicolor*), house wren (*Troglodytes aedon*), and American robin (*Turdus migratorius*)). Contaminant concentration of egg given in terms of lipid or dry weight were converted to wet weight (mg/kg) using data on lipid or

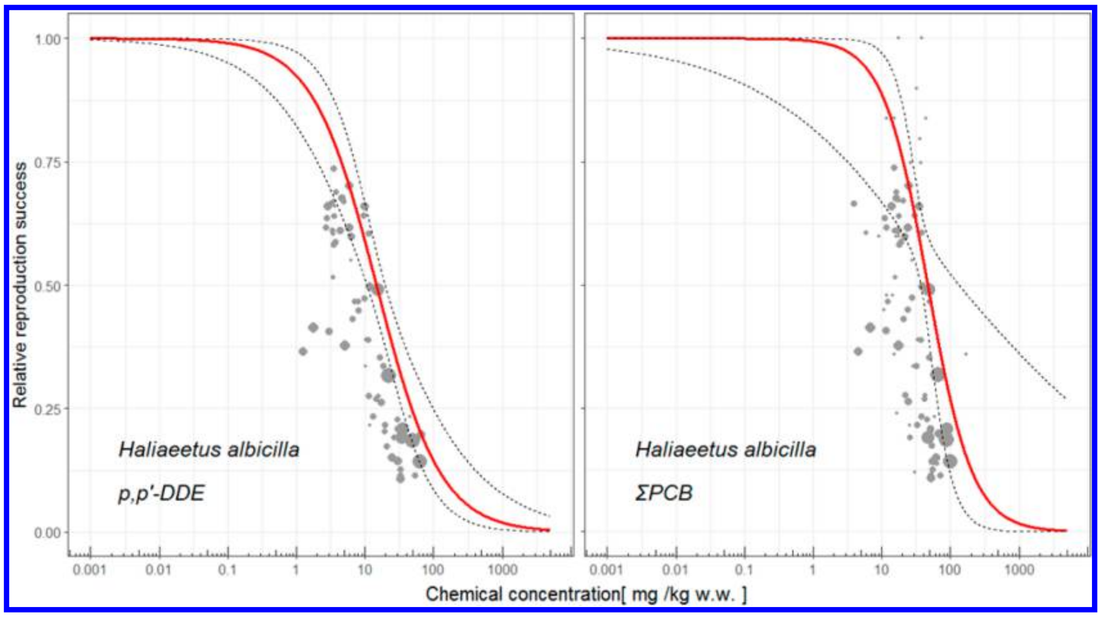

Figure 2. Quantile exposure-response curves plotting reduction in reproductive success at increasing chemical concentrations in eggs for the white-tailed eagle (*Haliaeetus albicilla*). Quantile τ was set at 0.95. The scatter points correspond with single data records, and their size corresponds with the assigned weight. The 95% C.I. is indicated through dashed lines. Exposure-response curves for the other birds species included can be found in the Supporting Information (Figures S1 and S2).

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Table 1. Effect Concentrations (mg/kg egg w.w.) Associated with the Derived Quantile Log-Logistic Regression Curves ($\tau = 0.95$) and Corresponding 95% Confidence Interval

	$\tau = 0.95$											
	p,p'-DDE						ΣPCBs					
species	N	β (±SE)	EC_{50}	95-CI	EC_{10}	95-CI	N	β (±SE)	EC_{50}	95-CI	EC_{10}	95-CI
Accipiter nisus[a]	54	−5.03 (±0.508)	27.5	19−66.3	17.8	12.1−32.9	54	−5.24 (±0.295)	28.6	10.1−∞	18.8	8.98−∞
Egretta thula[b]	12	−6.64 (±0.986)	7.65[d]	3.1−∞	5.5[d]	2.3−∞	6	−0.434 (±0.0368)	4.62	2.09−69.8	0.0296	0−0.148
Falco sp.[a]	37	−2.52 (±0.089)	16.9	13.9−∞	7.1	0.023−9.96	37	−1.64 (±0.0638)	27.1	17.7−348	7.1	0.014−12.8
Haliaeetus albicilla[a]	68	−0.93 (±0.0257)	14.7	11.1−18.5	1.38	0.276−2.7	88	−1.33 (±0.0381)	46.8	37.3−92.5	8.97	0.0229−17.3
Haliaeetus leucocephalus[a]	52	−0.428 (±0.0219)	36.9	12−67	0.218	0−0.92	50	−0.693 (±0.0258)	36.2	21.5−56.3	1.52	0.01−8.81
Larus argentatus[b]	28	−5.21 (±0.771)	16.4	10.9−∞	10.8	8.23−∞	28	−5.53 (±0.285)	124[d]	92−∞	83.2[d]	0.746−∞
Nycticorax nycticorax[b]	27	−3.66 (±0.145)	24.8	5.4−∞	13.6	2.01−∞						
Pandion haliaetus[b]	48	−0.778 (±0.0895)	4.58	0−∞	0.272	0−∞	44	−1.93 (±0.074)	8.5	0.536−9.73	2.72	0−4.36
Pelecanus occidentalis[b]	19	−4.64 (±0.137)	15.2	4.6−∞	9.47	1.62−∞	14	−19.2 (±0.754)	7.67[d]	5.1−∞	6.84[d]	0−14
Phalacrocorax sp.[b]	22	−3.66 (±0.239)	8.15[d]	1.5−∞	4.47[d]	0.183−∞	33	−1.45 (±0.0125)	14.8[d]	2.77−∞	3.3[d]	0−∞
Rynchops niger[b]							11	−9.02 (±0.691)	4.72[d]	1.43−∞	3.7[d]	1.76−∞
Sterna hirundo[b]	6	−11.2 (±2.3)	9.41	6.66−10.4	7.73	0−16.4	16	−2.77 (±0.062)	55.9	5.1−∞	25.3	1.18−∞
Tachycineta bicolor[c]	8	−1.25 (±0.0503)	8	0.9−∞	1.38	0−∞	21	−0.934 (±0.0153)	24.4[d]	3.35−∞	2.32[d]	0−10.5
Troglodytes aedon[c]							4	−0.619 (±0.164)	10.8[d]	0−∞	0.31[d]	0−∞
Turdus migratorius[c]							6	−3.75 (±1.15)	0.233	0.0032−14.4	0.13	0.14−38.9

[a]Raptorial bird. [b]Piscivorous bird. [c]Insectivorous bird. Additionally, 95% confidence intervals associated with the EC_{50}s/EC_{10}s are given. [d]Excluded from SSD derivation according to set conditions.

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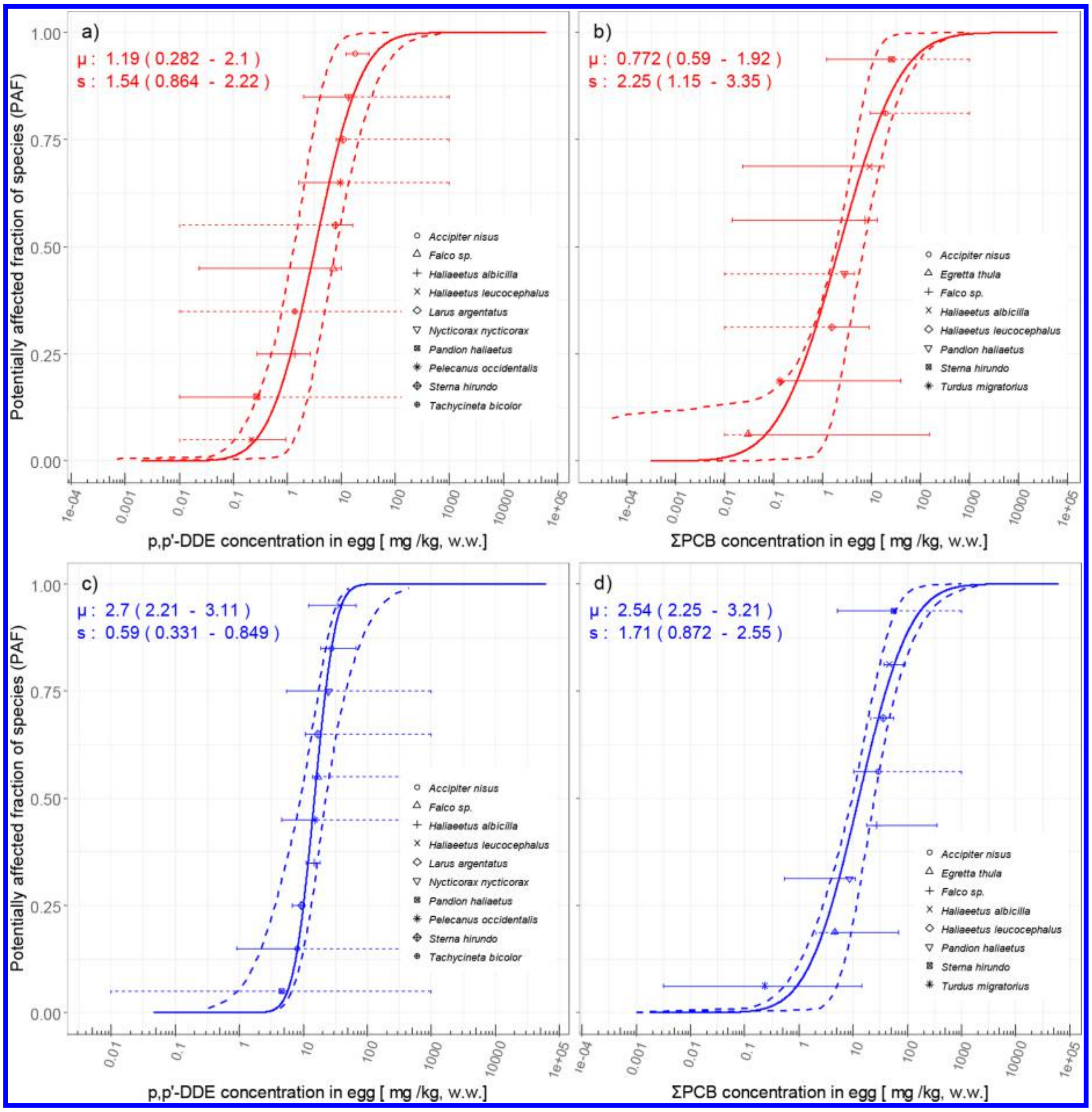

Figure 3. Species sensitivity distributions for derived from field-based EC_{10}s (in red) and EC_{50}s (in blue) for *p,p′*-DDE (a, c) and ΣPCB (b, d), respectively. Error bars around each $EC_{10/50}$ point indicate its 95% confidence intervals (dashed error bars indicate infinite confidence intervals). 95% confidence intervals corresponding to the derived SSDs are indicated as dashed lines. Additionally, the natural log-transformed μ and σ (s) per SSD was given (95% confidence interval between brackets).

moist percentage, respectively. The specific composition of PCB congeners included in ΣPCBs is listed per source in Table S2 of the Supporting Information.

Weights. The data set used in this study consists of both data obtained from single nests, as well as averaged bird productivity and egg residue concentrations, encompassing multiple nests, years, or locations. Therefore, in order for a larger sample size to yield a large influence on the analysis, the data points relating toxicant concentrations in eggs to reproductive success were weighted according to eq 8

$$w_i = \frac{1}{\frac{1}{N_{\text{nests}}} + \frac{1}{N_{\text{eggs}}}} \tag{8}$$

where w_i defines the weighting factor corresponding to data point i, N_{eggs} refers to the number of sampled eggs per toxicant data record, and N_{nests} refers to the number of active nests per productivity data record. With eq 8, we valued the information required for the productivity (based on the number of nests sampled) and chemical residues (based on the number of analyzed eggs) equally. This means that the weights are limited by the smallest sample size.

Exposure Data Associated with Ecological Risk. Additional data describing contaminant concentrations in bird's eggs used in calculating ecological risks by *p,p′*-DDE and PCBs were obtained using the Web of Knowledge in searches combining the terms *egg residues*, *birds*, and multiple specific areas. This search resulted in contamination data for two distinct areas (the Arctic[87−101] and the American Great Lakes[102−110]). Single toxicant exposure data were natural log-transformed and used to construct a probability density distribution that was applied in the calculation of ecological risks. Exposure data were

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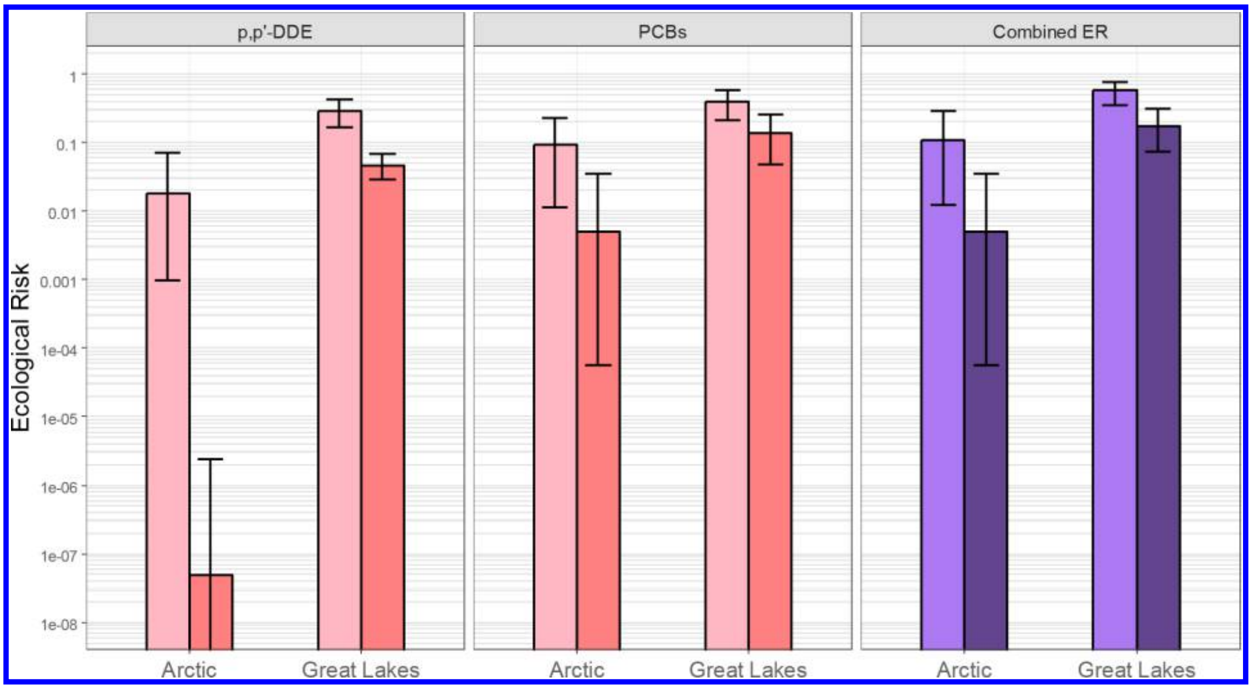

Figure 4. Ecological risks [fraction] with corresponding 95% confidence intervals in the Arctic and the Great lakes for PCBs, *p,p*-DDE, and their combined risk, based on bird EC_{10}s (light) and EC_{50}s (dark).

included without further weighting of number of eggs and nests.

■ RESULTS

Quantile Exposure-Response Curves. Quantile exposure-response curves, set at the 95th percentile, were fitted for 12 (*p,p′*-DDE) and 14 (ΣPCBs) raptorial, piscivorous, and insectivorous bird species for the two compounds as exemplified in Figure 2 for *Haliaeetus albicilla*. We derived 26 exposure-responses, as for 4 out of 15 species we only found sufficient data to include one of the two POPs concerned (see Figure S1 (*p,p′*-DDE) and Figure S2 (ΣPCBs) in the SI). Excluding species-substance combinations that were not considered sufficiently consistent with the other quantiles resulted in 18 exposure-response curves covering 12 species (see Figures S3 (EC_{50}s) and S4 (EC_{10}s) in the SI).

A significant decrease of reproductive success in birds along increasing toxicant gradients was observed for both compounds for exposure-response curves set at the 95th percentile. EC_{50}s ranged from 4.58 mg/kg egg w.w. (*Pandion haliaetus*) to 37 mg/kg (*Haliaeetus leucocephalus*) for *p,p′*-DDE and from 4.62 mg/kg (*Egretta thula*) to 124 mg/kg (*Larus argentatus*) for ΣPCBs. EC_{10}s ranged from 0.2 mg/kg (*Haliaeetus leucocephalus*) to 17.7 mg/kg (*Accipiter nisus*) for *p,p′*-DDE and from 0.03 mg/kg (*Egretta thula*) to 83 mg/kg (*Pandion haliaetus*) for ΣPCBs (Table 1).

Species Sensitivity Distributions. SSD models were derived from the field-based effect concentrations for the two contaminants, based on 18 quantile exposure-response curves set at the 95th percentile. As the quantile regression estimates in some cases provide zero or infinite EC_{50}- or EC_{10}-values due to the large statistical uncertainties involved, not all 10,000 realizations of the SSD curves yielded numerical results. For *p,p′*-DDE, only 9−13% of the iterations resulted in a numerical mean (μ) and standard deviation (σ) of the SSD, while for ΣPCB this percentage was higher with 32−37% of the iterations. This most likely results in an underestimation of the uncertainty bounds of the SSDs, as presented in Figure 3. Average toxicity (e^{μ}) based on EC_{50}s and EC_{10}s derived for *p,p′*-DDE is 14.9 mg/kg egg w.w. (9.1−22.4) and 3.3 mg/kg (2.9−8.2), respectively. Average toxicity associated with ΣPCB contamination is 12.7 mg/kg egg w.w. (9.5−24.8) and 2.2 mg/kg (2.1−6.8) based on EC_{50}s and EC_{10}s, respectively.

Ecological Risks. Single toxicant ERs calculated for the two compounds separately were a factor of 16 (*p,p′*-DDE) to 4.3 (ΣPCBs) higher in the North American Great Lakes compared to Arctic regions, based on 10% response levels. Individual ERs for *p,p′*-DDE based on 50% response levels were 5 orders of magnitude (*p,p′*-DDE) higher in North American Great Lakes than in Arctic regions, while individual ERs for ΣPCBs were a factor of 27 higher in North American Great Lakes than in Arctic regions. Consequently, differences in the overall combined ER_{10}s calculated for the American Great Lakes and Arctic sites were shown to be statistically significant, calculating higher combined ER_{10}s for the American Great Lakes compared to Arctic sites (5.7×10^{-1} vs 1.1×10^{-1}, respectively). The same holds for the combined ER_{50}s (1.7×10^{-1} vs 4.9×10^{-3}). PCB-contamination contributed most to combined ERs in the American Great Lake District and the Arctic based on both ER_{10}s and ER_{50}s (Figure 4).

Of all the Great Lakes, the highest combined ER_{10} and ER_{50} were calculated for Lake Ontario (6.3×10^{-1} and 2.2×10^{-1}, respectively). In the Arctic region, the highest combined ER_{10}s and ER_{50}s were calculated for the Barents Sea (1.23×10^{-1} and 5.4×10^{-3}, respectively), Norton Sound (1.31×10^{-1} and 9.6×10^{-3}), and Bering Sea (1.2×10^{-1} and 7.42×10^{-3}) (see Table S5 in the Supporting Information).

■ DISCUSSION

Our study explained how field data can be used to systemically derive chemical-specific and combined ecological risks. We also showed how the method can be applied in practice for *p,p′*-

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DDE and ΣPCBs in the Arctic and the American Great Lakes. Below, we discuss the three key elements of our study, i.e. the derivation of exposure-response curves, species sensitivity distributions, and ecological risks, respectively.

Exposure-Response Curves. In the present study, we derived exposure-response curves for multiple piscivorous, raptorial, and insectivorous bird species, relating *p,p*′-DDE and ΣPCBs in eggs to reproductive success, based on field data. EC_{50}s obtained in this study were higher compared to field-based EC_{50}s calculated for the same bird species from other studies,[45,111−113] likely due to the fact that other field studies use traditional regression approaches that focus on mean effects.[14] Our exposure-response curves were fitted along the upper boundary of the response's distribution, resulting in higher EC_{50}s. Effect concentrations calculated using traditional regression analysis (τ = 0.50; Table S3 in the Supporting Information) in the present study were similar to (for *Falco* sp.,[111] *Pelecanus occidentalis*,[113] and *Pandion haliaetus*[113]) or lower (for *Haliaeetus albicilla/leucocephalus*[112]) than those derived in other field studies.

It should be noted, however, that our exposure-response curves resulted in excluding 6 out of 14 species for ΣPCB and 2 out of 12 species for *p,p*′-DDE due to inconsistencies with lower quantile estimates. We also obtained very large uncertainty intervals for a number of species, depending on the number and distribution of the field-based exposure response data. These uncertainties were further propagated to the SSDs and ERs. We were, however, not able to fully quantify the uncertainty intervals of the SSDs and ERs, as not all simulations converged to a numerical outcome. This finding emphasizes the importance of having sufficient and well-distributed field data to reliably perform quantile regression analysis.

Species Sensitivity Distributions. Typically, the utility of SSD models to predict toxicity effects on the ecosystem level depends on a number of assumptions.[114] First, it is typically assumed that a log-normal distribution describes species sensitivities toward a certain chemical[4] and that including ten or more species is preferable to obtain representative SSDs.[28] In our case, the toxicity data were indeed lognormally distributed, while the number of species included was eight to ten. Another important assumption is that species selection is unbiased and fully represents the differences in sensitivity toward a chemical.[4] In the present study, SSDs were based on raptorial, piscivorous, and insectivorous bird species, covering multiple taxonomic groups. Differences in sensitivity of bird species to dioxin-like compounds, such as certain PCB congeners, are, however, explained by differences in the aryl hydrocarbon receptor 1 ligand-binding domain (AHR1-LB) and not so much by food sources (see Table S6 in the Supporting Information). In the present study, *Larus argentatus* and *Sterna hirundo* were identified as the species with the highest EC_{50} for ΣPCBs, followed by *Haliaeetus* species and *Falco* sp.. These findings correspond with the classification of Farmahin et al.,[115] who indicated that these birds are insensitive to dioxin-like compounds. The species with the highest sensitivity toward ΣPCBs in our study was the insectivorous bird *Turdus migratorius*. Farmahin et al.[115] classified this bird as semisensitive to dioxin-like compounds. Other birds classified as semisensitive were *Tachycineta bicolor*, *Troglodytes aedon,* and *Phalacrocorax* sp. Although experimental studies focusing on the reproductive effects of PCB residues in eggs have been performed on species classified as highly sensitive (e.g., European starling (*Sturnus vulgaris*) or the gray catbird (*Dumetella carolinensis*)),[116,117] field monitoring data on these species was grossly lacking. This implies that our SSDs most likely underestimate the effects of dioxin-like PCBs toward birds.

Ecological Risks. Ecological risks were significantly higher in the North American Great Lakes compared to the Arctic region. This is most likely due to the high number of pollutant point sources located along the Niagara and Detroit Rivers and the massive storage of organochlorine compounds in lake sediments that are re-emitted into the water column and subsequently accumulate in the food chain.[118,119] Note, however, that the calculated ecological risks for the American Great Lakes do not necessarily cover the trophic levels evenly, as in this area we found egg residue data encompassing a limited set of species, covering mainly *Tachycineta bicolor* and *Larus argentatus*.

Contamination with PCBs contributed highly to our calculated combined ERs in both areas. While the field-based average toxicity and spread between the species is rather similar for the two chemicals included in our study, systematically higher ΣPCBs concentrations were reported in the field compared to *p,p*′-DDE. Within the Arctic region, the high relative ecological risk (and thus combined ER) caused by PCB-contamination is most likely due to point sources from landfill, drilling rigs, harbors, and urban areas in Alaska (Bering Sea and Norton Sound) and Northern Norway (Barents Sea).[120] ERs calculated in other Arctic areas were relatively low, possibly due to geographical remoteness and the lower anthropogenic stress that is associated with this. The relatively high combined ERs for the Great Lakes were due to high individual ERs related to both PCB and *p,p*′-DDE contamination. The highest combined ER in the American Great Lake area was calculated for Lake Ontario, most likely caused by intensive farming in its catchment basin,[121] followed by Lake Erie and Lake Huron. These findings are in line with conclusions drawn in previous studies focusing on chemical residues in sediment, surface water, and fish, indicating Lake Ontario as most contaminated.[122,123] Obviously, risks may be different for individual bird species in other areas or specific locations within the Arctic and Great Lakes, depending on the actual concentrations present and the sensitivity of the species. For instance, effects on *Pelecanus occidentalis* in South Carolina and *Phalacrocorax auritus* in Green Bay were associated with *p,p*′-DDE rather than PCBs.[124,125] By contrast, in Dutch colonies, multiple studies suggest that *Phalacrocorax carbo sinensis* survival and reproduction and subsequent population development could partly be explained by PCB concentrations.[71,126]

Relevancy. SSDs are typically derived using data obtained from laboratory experiments, including only a limited number of species.[15,116,127] The present study illustrates how bird species monitoring data can be used to derive field-based SSDs. Although field-based SSDs are used before in ecotoxicology,[5,127,128] to our knowledge this study is the first to derive SSDs for piscivorous, raptorial, and insectivorous bird species that include effect concentrations obtained through quantile regression analysis that reduces bias associated with extraneous variables. SSDs derived from field monitoring data are considered more ecologically relevant. The use of field data is recommended by the U.S. Environmental Protection Agency Science Advisory Board for the derivation of ecological risks of chemicals.[129] This study demonstrates that the field-based

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approach can be applied to calculate ecological risks by combining SSDs with measured environmental POP concentrations.

ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.7b05904.

Figures S1 and S2, exposure-response curves obtained for p,p'-DDE and ΣPCBs at $\tau = 0.95$; Table S1, all maximum productivity values per species; Table S2, all PCB congeners used in the derivation of the exposure-response curves and the exposure concentration distribution for ΣPCBs listed per reference; Figures S3 and S4, all modeled effect concentrations at the $\tau = 0.95$ and the three additional quantiles ($\tau = 0.25$, $\tau = 0.5$, $\tau = 0.75$) for EC_{10} and EC_{50}, respectively; Table S3, EC_{10}s and EC_{50}s associated with traditional exposure-response curves ($\tau = 0.5$ and their 95% confidence intervals; Table S4, mean and standard deviations of $\log_{10}$-transformed egg concentrations found in the American Great Lake area and the Arctic used in calculating ERs; Table S5, all calculated ER_{10}s, ER_{50}s, and combined ERs per subregion; Table S6, all subtypes of AHR1-LB domains explaining sensitivity toward dioxin-like compounds of bird species used in this study (PDF)

R-code used in the construction of the quantile exposure-response curves (TXT)

Raw data that relates chemical contamination in egg residues to the productivity of bird species (XLSX)

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