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The influence of migration patterns on exposure to contaminants in Nearctic shorebirds: a historical study

Isabeau Pratte (ORCID) · **David G. Noble**
Mark L. Mallory (ORCID) · **Birgit M. Braune**
Jennifer F. Provencher (ORCID)

Abstract Since the 1970s, many populations of shorebirds, including those breeding in the Arctic region, have been declining. One factor that may contribute to some of these declines is exposure to contaminants throughout the annual cycle. Here, we compared contaminant exposure (organochlorines, toxic trace elements) of four Arctic-breeding shorebirds (semipalmated plover *Charadrius semipalmatus*, semipalmated sandpiper *Calidris pusilla*, lesser yellowlegs *Tringa flavipes*, and short-billed dowitcher *Limnodromus griseus*), collected during breeding, migration, and wintering to examine how and when contaminants might pose a threat to these species. In general, plovers and dowitchers had higher levels of most organochlorine pesticides, and renal cadmium (Cd) and selenium (Se) than the other species. Although we found seasonal differences, no clear patterns in contaminant concentrations among sampling locations were detected but the concentrations found at the breeding grounds were always the highest for chlorinated pesticides and mercury (Hg). Our results suggest that birds migrating south are slowly depurating contaminant burdens, and that spring-migrating birds were exposed to primarily North American rather than Latin American contaminant sources at the time of sampling. We present these data collected in the 1990s to better interpret current-day trends, and potential contaminant exposure impacts on shorebird populations.

Keywords Metals · Shorebirds · Annual migration · Organochlorines

I. Pratte (✉) · J. F. Provencher
Canadian Wildlife Service, Environment and Climate Change Canada, Place Vincent Massey, 351 St. Joseph Blvd, Hull, Quebec K1A 0H3, Canada
e-mail: pratteisabeau@gmail.com

D. G. Noble
British Trust for Ornithology, The Nunnery, Thetford, Norfolk IP24 2PU, UK

M. L. Mallory
Biology, Acadia University, 33 Westwood Drive, Wolfville, Nova Scotia B4P 2R6, Canada

B. M. Braune
Environment and Climate Change Canada, National Wildlife Research Centre, Carleton University, Raven Road, Ottawa, Ontario K1A 0H3, Canada

Introduction

At a global scale, shorebird populations have shown a rapid decline over the last several decades, attributed mainly to loss of breeding or stopover habitats (e.g., Pearce-Higgins et al. 2017; Amano et al. 2010). In Canada, 19 (42%) of 45 shorebirds, including many Nearctic species, showed declines in some parts of their range between 1974 and 1991 (Howe et al. 1989; Morrison et al. 1994). Although declines in some species have stabilized (Andres et al. 2012), more recent data indicate declines still prevail in most populations (Bart et al. 2007; Gratto-Trevor et al. 2011; Andres et al. 2012; Morrison et al. 2012; North

American Bird Conservation Initiative Canada 2019). Habitat loss and degradation on the breeding grounds and along migratory pathways are important factors contributing to those declines (Morrison et al. 1994; Page et al. 1999; Sutherland et al. 2012). Increased predation and habitat change on the breeding grounds (Lamarre et al. 2017; Flemming et al. 2019), hunting pressure on the wintering areas (Watts et al. 2015), and contaminant exposure throughout their range (Hargreaves et al. 2010; Saalfeld et al. 2016; Tsipoura et al. 2017) may also play additive/synergistic roles in contributing to the overall decline.

During the non-breeding season, most shorebirds feed on marine and aquatic invertebrates captured by gleaning from the surface of mud, water, and emergent vegetation, and by probing moist substrates (Skagen and Oman 1996). Consequently, shorebirds often feed in potentially contaminated areas such as estuaries (Burger et al. 1992) and flooded agricultural fields (Dias et al. 2014). Sediments and soils, some of which may be contaminated, are often ingested during feeding along with invertebrates (Hui and Beyer 1998). Most of the ~40 shorebird species that breed in Canada are long-distance migrants (Morrison et al. 1994), and thus may be exposed to contaminants during migration which includes overwintering areas in the USA and Central/South America and stopover areas in Canada and the USA, as well as on their breeding grounds.

Although the USA and Canada restricted use of organochlorine pesticides such as DDT in the early 1970s, use in many Latin American countries continued into the 1980s and is still used today in agriculture fields and also for malaria control (Nicholls and Altieri 1995; Roberts et al. 1997). The toxic effects of organochlorine pesticides, such as DDT and dieldrin, and polychlorinated biphenyls (PCBs) have been extensively documented in marine and fish-eating birds (e.g., Furness and Camphuysen 1997; Letcher et al. 2010; see specific reviews in Beyer and Meador 2011), but little is known about levels or their impacts in shorebirds.

Recent studies examining the impacts of contaminants on shorebird species as a possible contributor to declining populations are often hampered by having limited historical data on shorebird contaminant levels (e.g., McCloskey et al. 2013). Thus, data on contaminant levels from historical samples are useful to interpret current-day trends and potential impacts on populations. Given the sensitivity of waterbirds to organochlorine contamination (Tanabe 2002), and considering the continuing declines of many Arctic-breeding shorebirds, there is a need to assess how factors such as contaminants may be negatively affecting populations. To specifically address knowledge gaps in relation to baseline shorebird contaminant levels, we present historical data for four shorebird species (semipalmated plover *Charadrius semipalmatus*, semipalmated sandpiper *Calidris pusilla*, lesser yellowlegs *Tringa flavipes*, and short-billed dowitcher *Limnodromus griseus*) sampled during the early 1990s from different sites along their migratory pathways between their breeding grounds in Canada to their wintering grounds as far south as South America. The main objective of this study was to assess exposure of shorebirds breeding at Cape Churchill in northern Canada to contaminants at different stages of their annual cycle: on the breeding grounds, at staging areas during spring and autumn migration, and while overwintering in the Neotropics. To achieve this, we explored seasonal patterns in contaminant exposure while considering potential interspecific differences. Differences in diet and foraging behavior, migratory routes, wintering areas, and physiological processes (i.e., detoxification mechanisms, metabolic rates, excretion, and molt) are likely to contribute to interspecific differences in contaminant levels among stages of the annual cycle. Notably, we expected that species foraging at a higher trophic level or having a broader diet would have higher contaminant levels because of inclusion of higher trophic level species in their diet. For example, semipalmated plovers select a large proportion of polychaete worms (95%; Nol and Blanken 2014) during the non-breeding season while these predatory worms are less important in semipalmated sandpiper and short-billed dowitcher diets (Hicklin and Gratto-Trevor 2010; Jehl et al. 2001). Lesser yellowlegs have a broad and diverse diet especially during migration (Tibbits and Moskoff 2014).

While the focal species were of interest from a conservation perspective, we specifically considered their extensive migratory patterns to examine trends in contaminant levels across their annual cycle. Shorebirds migrate through many jurisdictions of varying contamination levels. In addition, some shorebirds that undertake trans-oceanic flights feed at high rates in staging areas to build up the fat reserves necessary for prolonged flights to their wintering grounds (Pfister and Kasprzyk 1998; Petit et al. 2010; Piersma and van Gils 2011). As these lipids are metabolized, lipophilic compounds such as organochlorines are released into the blood and redistribu ted to other organs with potentially toxic effects (van den Brink et al. 1998; Bustnes et al. 2010). We thus expected lipid levels to vary by season and time at stopover sites, given the energetic demand associated with migration and the necessity to refuel energy reserves at both spring and autumn stopovers, or prior to migration, e.g., in late winter. Therefore, we predicted shorebirds would have higher lipid content during those refueling periods, and similarly higher levels of lipophilic contaminants (e.g., organochlorines such as DDT, dieldrin, HCHs, PCBs). Finally, given the differences in use and control of contaminants across the many regions these shorebirds traverse, we expected to observe higher contaminant levels in birds sampled during the portions of their migration outside of the Arctic, with a potentially very high load on their northbound spring journey (Delaware Bay) from their overwintering grounds in South America.

Material and methods

Focal species and collection sites

Four relatively abundant shorebird species with different bill lengths were chosen to represent different foraging strategies that might expose them to various pollutant sources: semipalmated plover (population stable on IUCN; hereafter referred to as plovers), semipalmated sandpiper (populations decreasing on IUCN; hereafter sandpipers), lesser yellowlegs (populations decreasing on IUCN; hereafter yellowlegs), and short-billed dowitcher (populations decreasing on IUCN; hereafter

dowitchers). Plovers are mainly surface gleaners, sandpipers and yellowlegs are surface gleaners and probers, and dowitchers probe deepest into the sediments (see species reviews in Jehl et al. 2001; Hicklin and Gratto-Trevor 2010; Nol and Blanken 2014; Tibbits and Moskoff 2014). All four species are highly migratory, breed in the North American Arctic, and overwinter along the north coast of South America. Sandpiper breeding in the eastern Arctic migrates to sites on the Atlantic coast, particularly the Bay of Fundy, in the autumn (Hicklin and Gratto-Trevor 2010), to build up fat stores for a 3-to-4-day trans-oceanic flight to wintering grounds in Central America and northern South America. Plovers and dowitchers migrate south to winter in areas from the southern USA to South America (Nol and Blanken 2014; Jehl et al. 2001). Although some fly overland, most plovers and dowitchers make a single trans-oceanic flight from the Atlantic coast to South America, the former continuing to move south as far as Argentina (Nol and Blanken 2014; Jehl et al. 2001). Yellowlegs winter throughout the southern USA and Central and South America (Tibbits and Moskoff 2014).

To assess exposure throughout the year, specimens were obtained from five different stages of their annual migratory cycle (Fig. 1): (i) breeding areas near Cape Churchill in northern Canada, (ii) autumn staging areas in the Bay of Fundy, on the east coast of Canada, (iii) late autumn arrival on wintering areas on the north coast of Venezuela, (iv) the same Venezuelan wintering areas in late winter prior to departure, and (v) a spring staging area on Delaware Bay, on the east coast of the USA. Sites at Cape Churchill and coastal Venezuela were selected to represent the broader breeding and wintering ranges of the focal species. The Bay of Fundy is the most important stopover site on the east coast of North America for autumn migrants, due to its high density of coastal invertebrate prey (Hicklin 1987; Gerwing et al. 2016) during the period of sample collection (1990–1993) whereas Delaware Bay was extremely important to spring migrants that fed on the eggs of horseshoe crabs (*Limulus polyphemus*) during those years (Myers 1986; Clark et al. 1993).

Specimens were collected by shotgun or caught in mistnets. Autumn collections in Venezuela were
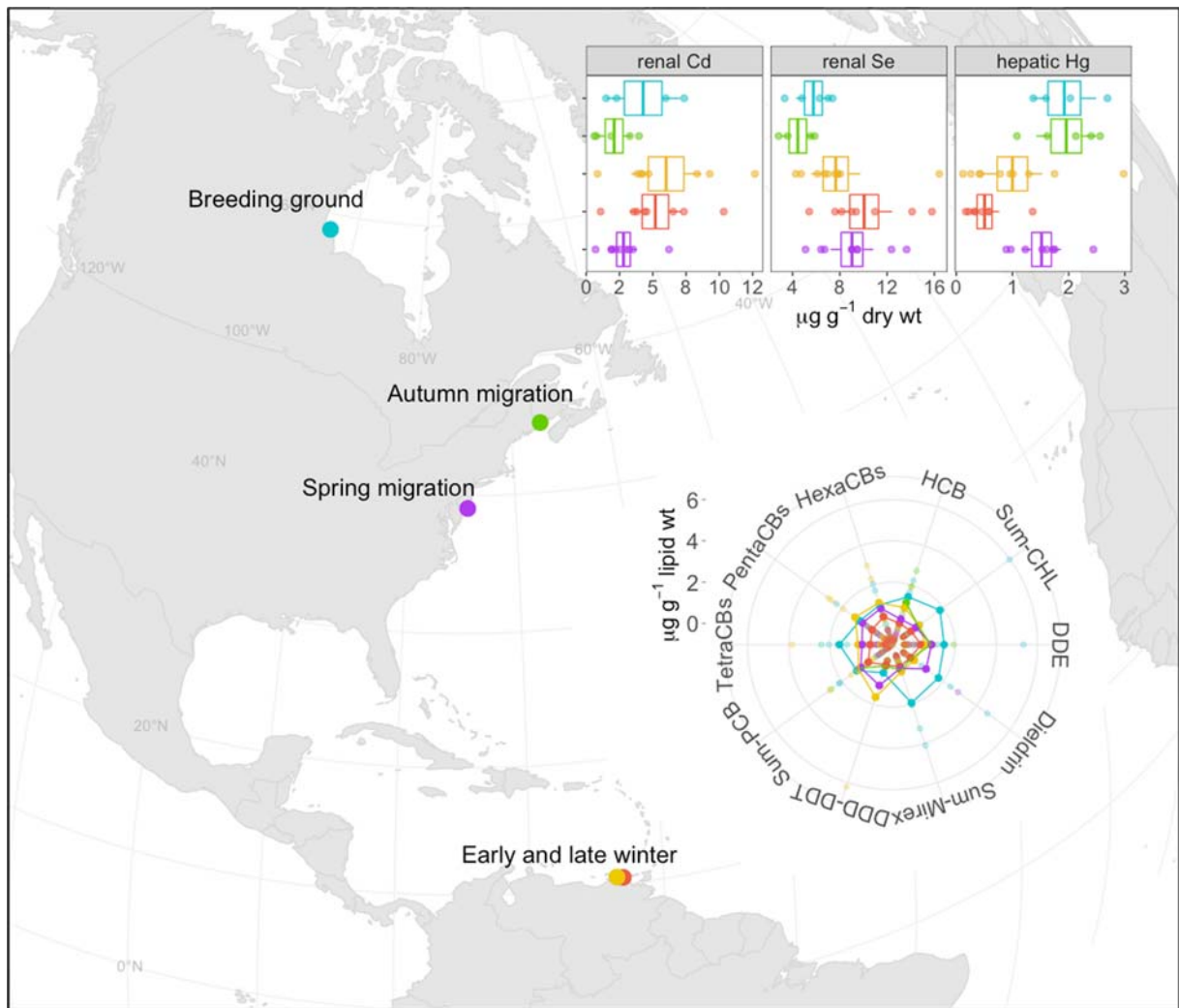
Fig. 1 Sampling locations of shorebirds collected along their annual life cycle in 1991 at Cape Churchill, CA - *Breeding grounds*; in 1991 in the Bay of Fundy, CA - *Autumn migration*; in 1992 and 1993 at the Araya Peninsula, VE - *Early winter* and *Late winter*; and in 1992 in Delaware Bay, USA - *Spring migration*. Mean (± standard error -box and 95% confidence interval - whiskers) concentrations (mg kg⁻¹ dry wt) of trace elements for which significant differences were found among sampling locations, and spider plot indicating mean organochlorine concentrations (scaled) for which significant differences were found among sampling locations. We omitted ΣHCH, oxychlordane and HE given their very low percent contribution (Fig. 3)

made between 8 and 22 September 1992 in coastal lagoons (Bocaripo and Chacopata) in the Araya Peninsula in the province of Sucre (11° 20′ N, 63° 45′ W). The late winter collections in Venezuela were made between 10 and 23 March 1993 at the same locations. Spring collections were made between 11 and 18 May 1992 at the Ted Harvey Conservation Area on Delaware Bay, between Delaware and New Jersey, USA (38° 56′ N, 74° 58′ W). The Bay of Fundy collections of plovers were made

in South Musquash, New Brunswick, Canada (45° 10′ N, 66° 16′ W) between 12 and 21 August 1991; sandpipers were collected between 30 July and 7 August 1991 in Passamaquoddy Bay (46° 06′ N, 66° 59′ W); dowitchers were collected between 21 and 26 July 1990 in Passamaquoddy Bay and Saints Rest Marsh (45° 13′ N, 66° 08′ W); and yellowlegs were collected between 23 August and 13 September 1990 and 10 October 1991 on Saints Rest Marsh. Collections on the breeding grounds were

made 12–28 June 1991 at Cape Churchill, Manitoba, Canada (58° 47′ N, 94° 11′ W); dowitchers and yellowlegs on or near tundra ponds, plovers along roads or tidal flats, and sandpipers on tidal flats. Details of these sites have been described elsewhere (e.g., Hicklin 1987; Clark et al. 1993). All collections were conducted under the authority of permits issued by the Canadian Wildlife Service (for Canadian samples), the Government of Venezuela (for Venezuelan samples), and the US Fish and Wildlife Service (for Delaware Bay samples). Samples were transported from Venezuela to Canada under Agriculture Canada's Permit to Import Animal Specimens and Products Samples (Permit No. 92-08-SPB-13) and Permit to Import Veterinary Biologics (Permit No. 252).

Prior to removal of the gut contents for a study of parasite loads (Didyk and Burt 1997; Didyk and Burt 1998a, 1998b), each bird was weighed to the nearest 0.5 g and (flattened) wingchord measured to the nearest 0.1 cm. Plumages of fresh birds were examined in the field for preliminary age and sex determinations, which were subsequently confirmed during dissection in the lab. Specimens were bagged and frozen as soon as possible after collection, within 4–8 h, and shipped to Environment Canada's National Wildlife Research Centre (NWRC) in Hull, Québec, for processing and residue analyses.

Legs, bill, liver, kidney, gastrointestinal tract, and a small amount of breast muscle were removed and the remaining carcass was defeathered. Samples were pooled on a species- and tissue-specific basis for chemical analyses and, hence, reported residue concentrations are approximately equivalent to arithmetic means for the samples in the pool (see Turle and Collins 1992). All early winter samples and one autumn sample (yellowlegs) were pooled by age class (hatch year, first year, adult), whereas spring, summer, and late winter samples (all adults) were pooled by sex. The eight adult male dowitchers collected from the Bay of Fundy were analyzed as two pools of four birds each based on the two collection sites (Passamaquoddy Bay and Saints Rest Marsh). Pooled samples were created by taking equal aliquots from each bird. We estimated variation in condition (fat level) based on variation in weights, or from the ratios of weight to wing length. This was used to assess the homogeneity of the pooled samples with respect to body condition as well as age and sex, and also to divide individuals into pools, as in the case of the

dowitchers from the Bay of Fundy. This was done to assess the homogeneity of the pooled samples with respect to conditions as well as age and sex. Because shorebirds undergo significant changes in body condition during migration (Hicklin 1987; Pfister and Kasprzyk 1998; O'Reilly and Wingfield 2003), specimens with similar fat levels were assumed most likely to have experienced similar periods of exposure on the collection site.

Chemical analyses

We analyzed 41 sample pools (189 individuals) of shorebird carcasses for organochlorines, which included females, males, and mixed-gender pools from between 1 and 9 individuals per pool (Appendix Table 4). Analytes included chlorobenzenes (1,2,4,5-tetrachlorobenzene, pentachlorobenzene and hexachlorobenzene), hexachlorocyclohexanes ($\Sigma HCH = \alpha$-, β-, and γ-HCH), chlordane-related compounds (ΣCHL = oxychlordane, trans-chlordane, cis-chlordane, trans-nonachlor, cis-nonachlor, and heptachlor epoxide), DDT and its metabolites (p,p'-DDE, p,p'-DDD, and p,p'-DDT), mirex (ΣMirex = photomirex and mirex), dieldrin, and PCBs, as Aroclor 1254:1260 and on a congener-specific basis (ΣPCB = 45 congeners (42 peaks) identified according to IUPAC numbers (Ballschmiter and Zell 1980): 28, 31, 42, 44, 49, 52, 60, 64, 66/95, 70, 74, 87, 97, 99, 101, 105, 110, 118, 128, 129, 137, 138, 141, 146, 149, 151, 153, 158, 170/190, 171, 172, 174, 180, 182/187, 183, 185, 194, 195, 200, 201, 203, and 206). Analysis of samples for organochlorines was carried out at NWRC using gas chromatography with electron capture detector (GC/ECD) (Norstrom and Won 1985). The value for the 1:1 ratio of Aroclor 1254 and 1260 is based on the peak for PCB congener 138. Blanks and NWRC in-house herring gull egg reference material (see Wakeford and Turle 1997) were run for quality assurance. The nominal detection limit for all compounds was 0.0001 $\mu g\ g^{-1}$ wet weight. Residues were not corrected for internal standard recoveries. Typical internal standard recoveries varied between 75 and 95% for most PCBs and organochlorines, and over 60% for the highly volatile compounds (i.e., chlorobenzenes). Lipids were determined by gravimetric methods.

Thirty-four kidney pools (165 birds) were analyzed for cadmium (Cd); 33 kidney pools (157 birds) for selenium (Se); and 34 liver pools (165 birds) for total mercury (Hg), again including females, males, and

mixed-gender pools from between 1 and 9 individuals (Appendix Table 5). We intended to analyze for lead but tissues showed strong evidence of contamination from lead shot. Organ samples were freeze-dried prior to trace element assays and digested in mineral acids (Neugebauer et al. 2000). Cd was analyzed by flame atomic absorption spectrophotometry (AAS). Se was analyzed by graphite furnace AAS. Total Hg was determined by cold vapor AAS. These methods are described in Neugebauer et al. (2000) and follow the method describe by Braune and Noble (2009). Analytical accuracy for each methodology was determined using blank samples and two standard reference materials (DOLT-2 and DORM-1 obtained from the Canadian National Research Council) as well as random samples run in triplicate.

Data handling and statistical analyses

Organochlorine concentrations are reported on a lipid-weight basis and metal concentrations on a dry-weight basis. Percent lipid data for tissues give an indication of the average condition of the birds in the pool. Only chlorinated pesticides (or groups) that occurred in more than 50% of the samples are included in the summary tables. These are p,p'-DDE, p,p'-DDD + p,p'-DDT, dieldrin, heptachlor epoxide (HE), oxychlordane, total chlordane (ΣCHL), total mirex (ΣMirex = photomirex and mirex), total HCH (ΣHCH = α-, β-, and γ-HCH), and hexachlorobenzene (HCB). For PCBs, concentrations are presented as the sum of all congeners (Σ_{45}PCB) and as Aroclor 1254:1260 for comparison with previous studies. Along with those, we compared concentrations of PCB homolog groups. However, trichlorobiphenyls (represented in these species only by very small amounts of congener 28, mainly in samples from Delaware Bay) are excluded from the comparisons. Nonetheless, we present all PCB homolog groups as percent composition for comparison with other studies.

One-half the detection limit (0.00005 µg g^{-1} wet weight) was assigned to non-detected values before data were log-transformed and tested for interspecific and seasonal effects using Kruskal-Wallis non-parametric rank sum test. Where significant differences occurred ($\alpha \leq 0.05$), differences between samples were tested using Dunn-Bonferroni post hoc comparisons of rank sums. All ages and sexes were combined for these analyses given the low sample size of pools. The analyses were carried out on dry weight for trace elements, and lipid-normalized (lipid-weight) data for organochlorines given the wide range of lipids found in the samples. All statistical analyses were carried out in R version 3.5.2 (R Core Team 2018).

Results

Lipid levels in carcasses analyzed for organochlorines

We found a significant difference among seasons in lipid levels ($\chi^2 = 14.12$, $p = 0.007$; Fig. 2). Dunn-Bonferroni post hoc comparison test indicated that birds sampled during both spring and fall migration had higher lipid levels than at the breeding ground (both $p = 0.007$). During spring migration, the only yellowlegs sampled had higher lipids (27%, $n_1 = 1$) than all other pool analyzed (Appendix Table 4). The opposite was found during autumn migration, where yellowlegs (8.8 ± 7.4, $n_{1,7} = 2$) had generally lower lipid levels than plovers (26.6 ± 7.5%, $n_{5,4} = 2$) and sandpipers (19.1 ± 3.8, $n_{5,5} = 2$). Dowitchers (16.1 ± 5%, $n_{5,5} = 2$) had seemingly lower lipid values than plovers for autumn migration. During this southbound stopover,
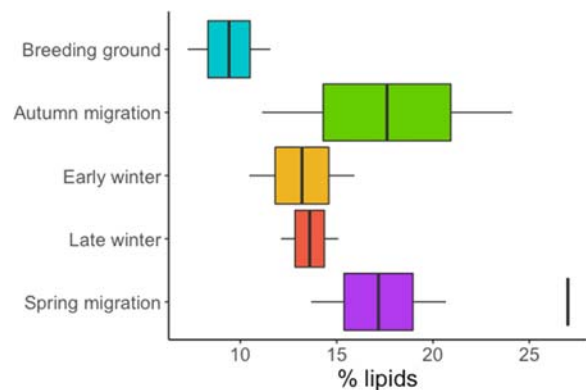

Fig. 2 Mean (± standard error -box and 95%confidence interval - whiskers) lipid levels (%) Lipid levels (%) in carcasses assessed for organochlorines of four species of shorebirds (lesser yellowlegs, semipalmated plovers, semipalmated sandpipers, short-billed dowitchers) collected along their annual life cycle in 1991 at Cape Churchill, CA - *Breeding grounds*; in 1991 in the Bay of Fundy, CA - *Autumn migration*; in 1992 and 1993 at the Araya Peninsula, VE - *Early winter* and *Late winter*; and in 1992 in Delaware Bay, USA - *Spring migration*

lipid levels in the shorebird pools varied from 1.4 to 33.7% with the highest from the pool of female semipalmated plovers, and the lowest in a single very lean adult lesser yellowlegs (Appendix Table 4). The overall wide range of lipid levels during autumn stopover in all species (Appendix Table 4) suggests a mix of lean and fat birds, i.e., new arrivals (lean) and those ready to depart (fat). Despite those apparent differences, we did not find that levels of lipids varied by species ($p = 0.42$). Because of substantial variation through the year in lipid levels, we report all contaminant comparisons in lipid-normalized concentrations (lw).

Levels and trends in organochlorine contaminants

The primary organochlorines found in shorebird carcasses were p,p'-DDE and $\sum_{45}$PCB although proportions varied by species and seasons (Fig. 3). Semipalmated sandpipers were mostly contaminated with $\sum_{45}$PCB and p,p'-DDE was less important except during spring migration. Dieldrin contribution was elevated in yellowlegs during spring migration compared with all other species and seasons. DDD and DDT were mostly present in semipalmated plovers during early winter in Venezuela (Fig. 3).

P,p'-DDE

DDE was detected in all pooled samples (Appendix Table 4), and the highest concentrations were found in female yellowlegs from the breeding grounds with 29.2 µg g^{-1} (Appendix Table 4). Significant interspecific differences were detected for p,p'-DDE ($\chi^2 = 16.020$; $p = 0.001$) (Table 1). Dunn-Bonferroni post hoc comparison revealed differences in p,p'-DDE due to the lower overall concentrations in sandpipers compared with the other species ($p < 0.002$; Table 3). DDE also varied according to season ($\chi^2 = 9.94$; $p = 0.04$) notably to higher concentration on the breeding grounds compared with early and late winter samples (both $p < 0.01$; Fig. 4).

PCBs

PCBs were detected in all sample pools, and the most prevalent homolog groups were hexa- and heptachlorobiphenyls (Fig. 5). PCB concentrations are presented as both the sum of PCB congeners ($\sum$PCB—total of 45 congeners) and as Aroclor 1254:1260 for comparison with values in the older literature (Appendix Table 4). $\sum_{45}$PCB differed significantly across seasons ($\chi^2 = 11.55$; $p = 0.02$; Table 1), with lower concentrations in late winter compared with the other seasons (all $p < 0.01$; Fig. 4). There were significant interspecific differences only for hexa-, hepta-, octa-, and nona-chlorobiphenyls (all $\chi^2 > 7.83$; $p < 0.05$; Table 1). Tetra-, penta-, and hexa-chlorobiphenyls exhibited significant seasonal differences (all $\chi^2 > 9.32$; $p < 0.05$; Fig. 4).

∑Mirex

Mirex and photomirex (the latter generally less prevalent) occurred in all samples except one (the pool of male short-billed dowitchers from the Bay of Fundy). ∑Mirex occurred from non-detectable to 0.66 µg g^{-1} in male plovers from the breeding grounds (Appendix Table 4). ∑Mirex varied across seasons ($\chi^2 = 18.90$; $p < 0.001$), shorebirds sampled at the breeding grounds had higher concentrations compared with the other seasons (all $p < 0.05$).

Dieldrin

Dieldrin occurred at concentrations < 0.11 µg g^{-1} except in male lesser yellowlegs at Delaware Bay during spring migration (1.36 µg g^{-1}), and female and male semipalmated plovers at Churchill on the breeding grounds (respectively 1.16 and 2.07 µg g^{-1}; Appendix Table 4). Dieldrin varied across seasons ($\chi^2 = 18.84$; $p < 0.001$), with higher levels found during early winter, spring migration and at the breeding ground (all $p < 0.05$; Fig. 4). No differences among species were found ($p = 0.40$; Table 1).

Oxychlordane and ∑chlordanes

Oxychlordane was highest in male plovers during autumn migration in the Bay of Fundy at 0.19 µg g^{-1} followed by male dowitchers from the breeding grounds with 0.38 µg g^{-1} (Appendix Table 4). In general, plovers appeared to have higher oxychlordane levels than the other
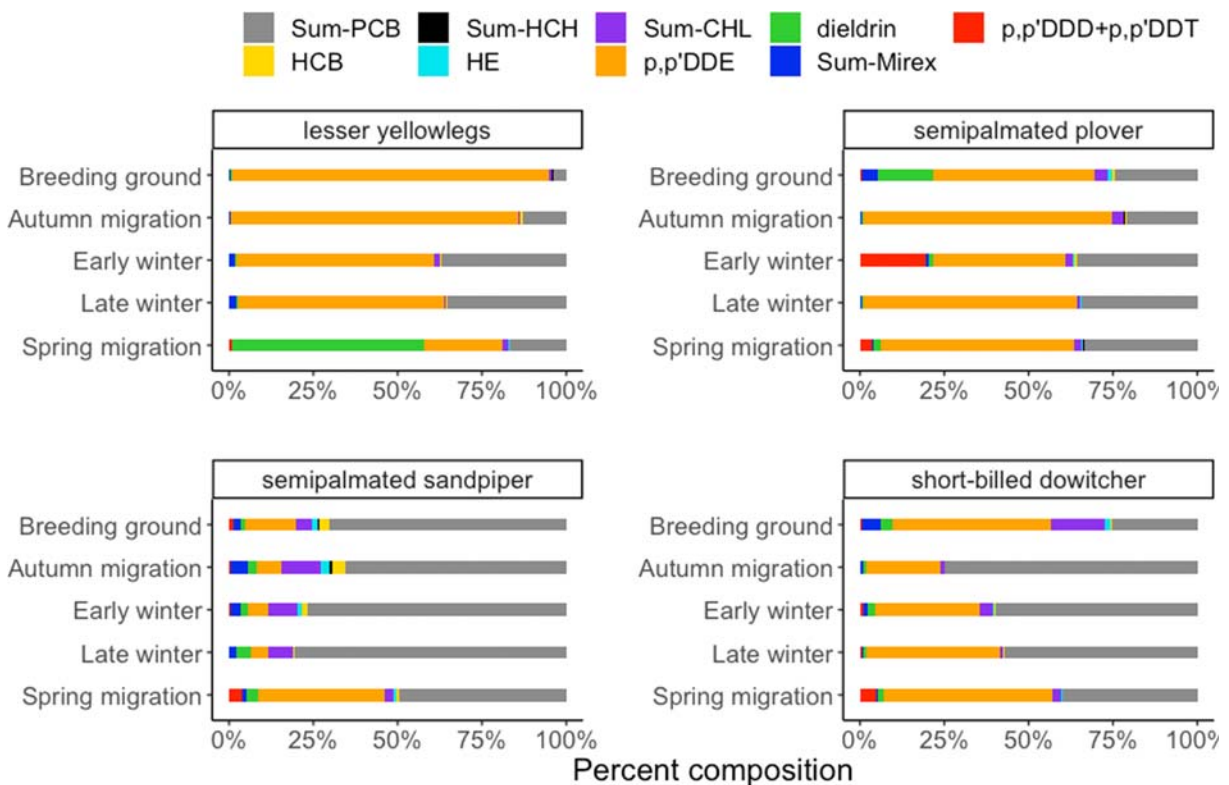
Fig. 3 Percent contribution of major organochlorine compounds/ groups to total organochlorines (∑OC) in four shorebird species collected along their annual life cycle in 1991 at Cape Churchill, CA - *Breeding grounds*; in 1991 in the Bay of Fundy, CA - *Autumn migration*; in 1992 and 1993 at the Araya Peninsula, VE - *Early winter* and *Late winter*; and in 1992 in Delaware Bay, USA - *Spring migration*

species, especially on the breeding grounds, although no species effects were detected for this residue (Table 1). In dowitchers, one sample from Churchill contained elevated residues of ∑CHL; however, no interspecific differences were found for the residue ($p = 0.35$). Both oxychlordane and ∑CHL varied according to season ($\chi^2 > 10.87$; $p < 0.02$; Table 1) again with higher levels found on the breeding grounds compared with other sites (all $p < 0.04$; Table 2), except similar levels of oxychlordane between breeding ground and early winter samples (Fig. 4).

P,p'-DDT and p,p'-DDD

DDT and DDD were detected at much lower concentrations than *p,p'*-DDE in about half of the pools (Appendix Table 4), and showed no

significant variation across species ($p = 0.49$; Table 1). The highest concentration was detected in early winter in Venezuela (3.81 μg g^{-1}; Appendix Table 4), but residue remained undetected in late winter at the same location suggesting potentially high individual variation. Seasonal effect was significant ($\chi^2 = 20.18$; $p < 0.001$), mostly a result of higher levels during spring migration and on the breeding ground compared with the other seasons (all $p < 0.03$).

Heptachlor epoxide

Residues of heptachlor epoxide (HE) varied from non-detectable to 0.23 μg g^{-1} in breeding male dowitchers (Appendix Table 4). HE residues varied only seasonally ($\chi^2 = 19.43$; $p < 0.001$; Table 1) and were higher at the breeding grounds

Table 1 Interspecific and seasonal differences in concentrations of pollutants in shorebirds. Results of Kruskal-Wallis test on lipid wt (dry wt for trace elements) concentrations (sexes and ages pooled)

Compound	Interspecific differences			Seasonal differences	
	χ^2	p		χ^2	P
Organochlorines					
HCB	5.58	0.13		13.78	0.008**
ΣHCH	2.60	0.46		15.69	0.003**
Oxychlordane	6.76	0.08		10.87	0.02*
HE	1.96	0.58		19.43	< 0.001***
ΣCHL	3.23	0.36		16.24	0.003**
p,p'-DDE	16.20	0.001**		9.94	0.04*
p,p'-DDD + p,p'-DDT	2.45	0.49		20.18	< 0.001***
Dieldrin	2.96	0.40		18.84	< 0.001***
ΣMirex	0.07	0.99		18.90	< 0.001***
Σ_{45}PCB	5.80	0.12		11.55	0.02*
Aroclor 1254:1260	5.56	0.14		6.77	0.15
-Chlorobiphenyls					
Tetra	1.67	0.61		17.50	0.001**
Penta	3.81	0.28		11.06	0.03*
Hexa	7.83	0.05*		9.32	0.05*
Hepta	10.55	0.01*		5.79	0.22
Octa	12.87	0.005**		5.53	0.24
Nona	10.94	0.01*		8.28	0.08
Trace elements					
Cadmium (renal)	13.34	0.004**		10.55	0.03*
Selenium (renal)	13.08	0.004**		14.23	0.006**
Mercury (hepatic)	1.98	0.58		16.80	0.002**

* $p \leqq 0.05$

** $p < 0.01$

*** $p < 0.001$

compared with all other sampling locations (all $p < 0.02$; Fig. 4).

Hexachlorobenzene

Hexachlorobenzene (HCB) occurred in 90% of samples at levels ranging from non-detectable to 0.07 μg g^{-1} (lw) in adult yellowlegs from the Bay of Fundy and early winter in Venezuela, and male sandpipers at Cape Churchill and in the Bay of Fundy during fall migration (Appendix Table 4). Seasonal effects were detected for HCB concentrations (χ^2 = 13.78; p = 0.008; Table 1); residues of HCB were significantly higher on the breeding

grounds compared with the late winter site, and the spring stopover during migration in Delaware Bay (both $p < 0.03$; Fig. 4), and were higher during autumn migration compared with late winter (p = 0.005; Table 2).

Hexachlorocyclohexane

Hexachlorocyclohexane (HCH) isomers were found in only 34% of the samples, mainly from the breeding grounds at Cape Churchill as well as from the Bay of Fundy. β-HCH was the major HCH isomer detected although α-HCH was found in all plover and sandpiper samples from the
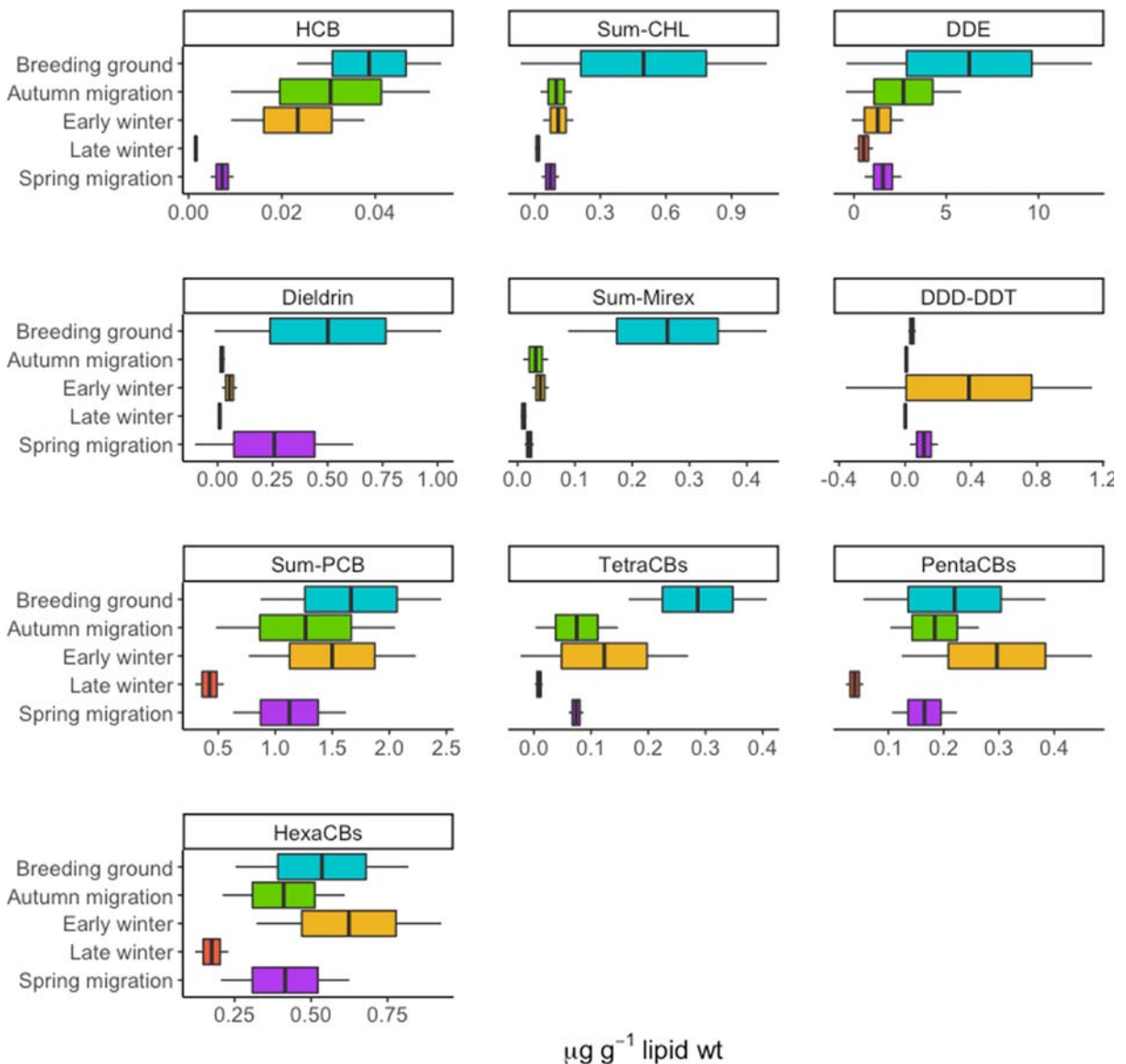
Fig. 4 Mean (± standard error -box and 95% confidence interval - whiskers) concentrations ($\mu g\ g^{-1}$ lipid wt) in carcasses of organochlorines for which significant differences were found among seasons in the annual cycle of four shorebird species collected along their annual life cycle in 1991 at Cape Churchill, CA - *Breeding grounds*; in 1991 in the Bay of Fundy, CA - *Autumn migration*; in 1992 and 1993 at the Araya Peninsula, VE - *Early winter* and *Late winter*; and in 1992 in Delaware Bay, USA - *Spring migration*. We omitted ΣHCH, oxychlordane, and HE given their very low percent contribution (Fig. 3). Note the different scales

breeding grounds, and all three isomers (α-, β-, γ-HCH) were found in the plovers at Cape Churchill as well as the pool of adult males from the Bay of Fundy. The highest concentrations of ΣHCH occurred in male plovers from the Bay of Fundy (0.06 $\mu g\ g^{-1}$; Appendix Table 4). ΣHCH concentrations varied among seasons ($\chi^2 = 15.69$; $p = 0.003$); ΣHCH was higher at the
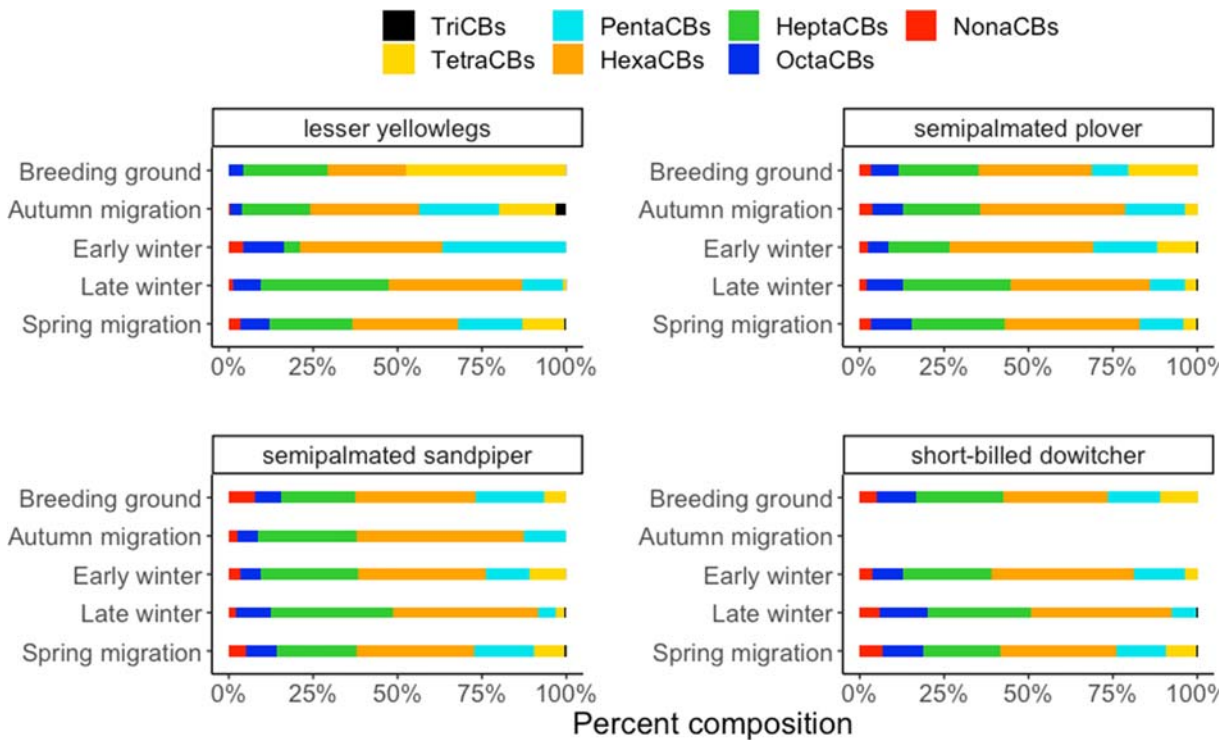
Fig. 5 Percent contribution of PCB homolog groups to Σ_{45}PCB in carcasses of four shorebird species collected along their annual life cycle in 1991 at Cape Churchill, CA - *Breeding grounds*; in 1991 in the Bay of Fundy, CA - *Autumn migration*; in 1992 and 1993 at the Araya Peninsula, VE - *Early winter* and *Late winter*; and in 1992 in Delaware Bay, USA - *Spring migration*. PCB congeners found in each homolog group are the following: TriCBs congeners 28, 31; TetraCBs congeners 42, 44, 49, 52, 60, 64, 66/95, 70, 74; PentaCBs congeners 87, 97, 99, 101, 105, 110, 118; HexaCBs congeners 128, 129, 137, 138, 141, 146, 149, 151, 153, 158; HeptaCBs congeners 170/190, 171, 172, 174, 180, 182/187, 183, 185; OctaCBs congeners 194, 195, 200, 201, 203; NonaCBs congeners 206

breeding grounds compared with all other seasons (all $p < 0.01$; Fig. 4).

Levels and trends in trace elements

Mercury in liver

Total Hg in livers varied over more than an order of magnitude from 0.12 µg g^{-1} (dw) in first-year dowitchers upon early winter arrival in Venezuela to almost 3.0 µg g^{-1} in hatch-year sandpipers also collected upon early winter arrival in Venezuela (Appendix Table 5). There were no significant interspecific trends, but concentrations varied significantly across seasons ($\chi^2 = 16.80$; $p = 0.002$; Table 1). Dunn-Bonferroni post hoc comparison test revealed this was due to a trend for higher mercury concentrations in shorebirds collected from northern locations (i.e., Delaware Bay, Bay

of Fundy, Churchill) compared with those in Venezuela, particularly in late winter (all $p < 0.05$; Fig. 6; Table 2).

Selenium in kidney

Renal Se concentrations varied sixfold from about 2.82 µg g^{-1} in yellowlegs from the Bay of Fundy to 16.4 µg g^{-1} in early winter adult plovers collected in Venezuela (Appendix Table 5). Samples of adults and immatures were collected concurrently during early winter (the four species) and also during autumn migration for yellowlegs. In all five cases, adults contained greater renal Se residues, but the differences were small except in plovers for which the difference was more than twofold. We found interspecific ($\chi^2 = 13.08$; $p = 0.004$) and seasonal ($\chi^2 = 14.23$; $p = 0.006$; Table 1) differences in renal Se concentrations.

Table 2 Seasonal differences in organochlorines (lipid wt basis) and trace element (dry-weight basis) concentrations. ($p \leq 0.05$) following Dunn-Bonferroni multiple comparisons post hoc test with a being the highest value. Only residues that significantly differed (*chi*-statistic $p \leq 0.05$) among seasons are presented

	Season					*Chi*-square
	Breeding grounds	Autumn migration	Early winter	Late winter	Spring migration	
HCB	a	ab	ab	c	bc	13.78**
ΣHCH	a	b	b	b	b	15.69**
Oxychlordane	a	b	a	b	b	10.87*
HE	a	bc	b	c	b	19.43***
ΣCHL	a	b	b	c	b	16.24**
p,p′-DDE	ab	abc	c	c	b	9.94*
p,p′-DDD + *p,p′*-DDT	a	b	b	b	a	20.18***
Dieldrin	a	b	a	b	a	18.84***
ΣMirex	a	b	b	c	bc	18.90***
Σ_{45}PCB	a	a	a	b	a	11.55*
TetraCBs	a	bc	b	c	ab	17.50**
PentaCBs	a	a	a	b	a	11.06*
HexaCBs	a	a	a	b	a	9.32*
Cd (renal)	ab	b	a	a	b	10.55*
Se (renal)	cd	d	bc	a	ab	14.23**
Hg (hepatic)	ab	a	b	c	ab	16.80**

* $p \leqq 0.05$

** $p < 0.01$

*** $p < 0.001$

Post hoc comparison tests revealed generally lower renal Se concentrations in yellowlegs compared with the other three species (all $p < 0.004$; Table 3). Opposite to what we found in hepatic Hg levels, concentrations of renal Se found in shorebirds collected from the breeding grounds at
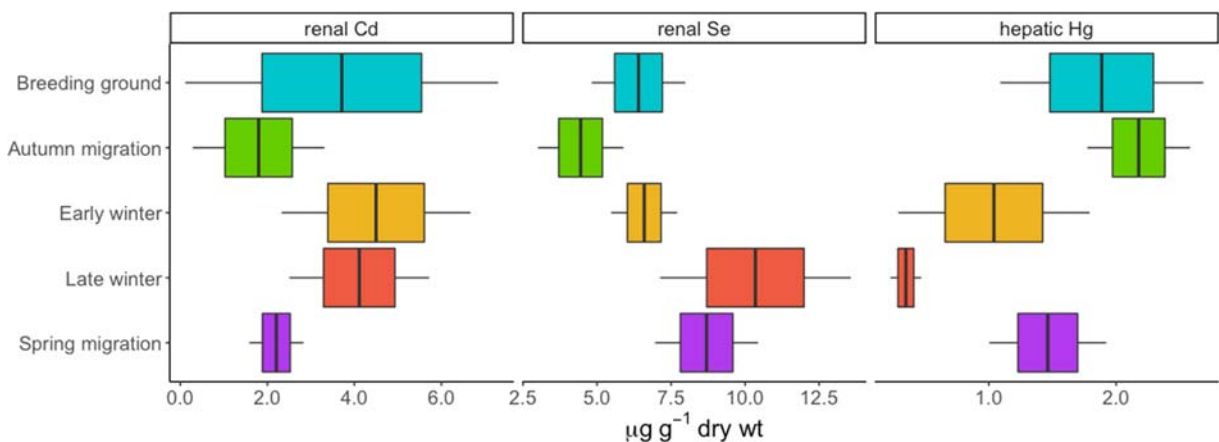
Fig. 6 Mean (± standard error -box and 95% confidence interval - whiskers) concentrations ($\mu g \ g^{-1}$ dry wt) of trace elements for which significant differences were found among seasons in the annual cycle of four shorebird species collected along their annual life cycle in 1991 at Cape Churchill, CA - *Breeding grounds*; in 1991 in the Bay of Fundy, CA - *Autumn migration*; in 1992 and 1993 at the Araya Peninsula, VE - *Early winter* and *Late winter*; and in 1992 in Delaware Bay, USA - *Spring migration*. Note that Cd and Se are concentration in kidney, and Hg is in liver, and the different scales

Table 3 Species differences in organochlorines (lipid wt basis) and trace element (dry-weight basis) concentrations. Different letters indicate significant differences ($p \leq 0.05$) following Dunn-Bonferroni multiple comparisons post hoc test with a being the highest value. Only residues that significantly differed (*chi*-statistic $p \leq 0.05$) among species are presented. SEPL semipalmated plover, SESA semipalmated sandpiper, LEYE lesser yellowlegs, SBDO short-billed dowitcher

	Species				Chi-square
	SEPL	SESA	LEYE	SBDO	
p,p'-DDE	a	b	a	a	16.20**
HexaCBs	a	b	b	ab	7.83*
HeptaCBs	a	bc	c	ab	10.55*
OctaCBs	a	b	b	a	12.87**
NonaCBs	a	a	b	a	10.94*
Cd (renal)	a	bc	c	ab	13.34**
Se (renal)	a	a	b	a	13.08**

* $p \leq 0.05$

** $p < 0.01$

Churchill, from the Bay of Fundy in the fall, and in early winter upon arrival at wintering ground were slightly lower than levels found in Venezuela in late winter (all $p < 0.04$; Fig. 6). As well, early winter shorebirds from Venezuela had higher Se levels than southbound migrating birds ($p = 0.03$; Table 2). Shorebirds collected during spring migration in Delaware Bay also had higher renal Se levels than birds sampled from Churchill and the Bay of Fundy (both $p < 0.02$; Fig. 6; Table 2), which suggest higher Se concentrations in birds collected from southernmost latitudes and onward northbound journey.

Cadmium in kidney

Renal Cd concentrations varied from trace in hatch-year plovers and yellowlegs to 12.7 µg g^{-1} in plovers collected in early winter (Appendix Table 5). Most samples contained < 5 µg g^{-1} (Appendix Table 5). We found significant interspecific differences ($\chi^2 = 13.34$; $p = 0.004$; Table 1) due to higher levels of renal Cd in plovers compared with yellowlegs and sandpipers (both $p < 0.01$; Table 3), and to slightly higher levels in dowitchers compared with yellowlegs ($p = 0.01$).

We also found differences among seasons ($\chi^2 = 10.55$; $p = 0.03$). Shorebirds collected in early and late winter had higher renal Cd concentrations than birds collected during spring and fall migration in Delaware Bay and in the Bay of Fundy (both $p = 0.01$; Fig. 6).

Discussion

Organochlorines

Despite the fact that most uses of organochlorine pesticides have been banned in North America since the early 1970s (Loganathan and Kannan 1994), although some use in Latin America has continued, a suite of chlorinated compounds and their metabolites, including DDE, dieldrin, heptachlor epoxide, mirex, HCH, and oxychlordane, were found in the shorebird carcasses sampled in the early 1990s. Organochlorine pesticides have high persistence in the environment, with some pesticide compounds like DDE, DDT, DDD, and chlordane having half-lives between 2 and 15 years (Jayaraj et al. 2016). Dieldrin and heptachlor, although highly persistent, have shorter half-lives of less than 2 years (Jayaraj et al. 2016). We also detected significant concentrations of polychlorinated biphenyls (PCBs) including 45 of the 209 possible PCB congeners. The sole North American producer of PCBs restricted sales since 1972, but PCBs continued to enter the environment via industrial effluent, combustion, illegal dumping, and leakage. Technical formulations of PCBs tend to be dominated by penta- and hexachlorinated congeners. Their persistence in the environment varies according to chemical properties but, in general, biodegradability decreases with increased chlorination (Tanaka et al. 1986). PCBs in the shorebird samples in this study were dominated by the hexa- and hepta-chlorinated congeners, suggesting that the persistence of these groups in the environment make them available to be uptaken by birds for decades. Other studies examining PCBs in Arctic marine bird species have also found the PCB profile to be dominated by penta-, hexa- and hepta-chlorinated congeners (Braune et al. 2018; Braune and Mallory 2017).

Interspecific differences

Consistent with our predictions based on diet, plovers had higher levels of most organochlorine pesticides than the other species. This was significant for plovers, yellowlegs, and dowitchers that had higher levels of p,p'-DDE compared with sandpipers; plovers and dowitchers had higher hexa-, hepta-, and octachlorinated congeners concentrations than sandpipers and/or yellowlegs. This is similar to what Johnstone et al. (1996) reported for Rankin Inlet, NU, between 1991 and 1994. They found higher levels of both DDE and PCB residues (0.50 and 0.40 $\mu g\ g^{-1}$ ww, respectively) in bodies of semipalmated plovers and lower levels in semipalmated sandpipers.

Due to our low sample size and use of nonparametric statistics, we could not test for interaction with species and site. Differences in residue levels could be reflecting differences in diet and physiological mechanisms for dealing with xenobiotics for each species depending on exposure and local contamination at the different sites. Within a species, differences in residue levels among sampling locations are expected to reflect local contamination, exposure prior to arrival on site and current physiological condition. At a particular site, interspecific differences in concentrations are likely to be most strongly influenced by differences in diet and physiological mechanisms for dealing with xenobiotics as well as exposure in the period prior to arrival on site. We acknowledge that a larger sample size would have enabled us to test whether or not contaminant exposure varied among species according to life history stages and how variation among life history stages might depend on the species considered; however, we could not acquire larger samples at that time. We briefly review possible causes of interspecific differences in contamination, which are discussed in greater detail and covering more species in a previous paper on environmental contaminants in shorebirds sampled across Canada (Braune and Noble 2009) which includes some of the data presented here.

Focal species were chosen to represent different foraging tactics. It was expected that surface gleaners such as plovers and yellowlegs were more likely to encounter atmospherically deposited pollutants than species that probe the sediments, such as dowitchers. However, some organochlorine compounds persist longer in less aerobic conditions deeper in the sediments (Young et al. 1988) and may be more readily available to deep probers. The latter hypothesis was not supported by Braune and Noble (2009) who did not find greater contaminant levels in longer-billed species that probe the sediment. The elevated organochlorine levels that we found in plovers support the atmospheric deposition hypothesis. However, levels of particularly volatile compounds such as HCH were not significantly higher in plovers than in other species, suggesting that a suite of factors influence contaminant uptake in a species.

For example, most shorebirds can use a range of foraging tactics and a range of microhabitats especially during migration to find varied prey items (Skagen and Oman 1996; Smith and Nol 2000; MacDonald et al. 2012; Quinn and Hamilton 2012; Smith et al. 2012; Gerwing et al. 2016). Most comparative studies of shorebird diet have been carried out on migration stopovers. Recent studies have shown considerable variation in diet of sandpipers in the Bay of Fundy, highlighting the importance of biofilm and polychaetes along with *Corophium volutator* (Quinn and Hamilton 2012). Such diet diversity and species similarities in diet could also take place during other periods of a shorebird's annual cycle, explaining the lack of species differences in most contaminants. Nonetheless, given that higher PCB congeners and pesticides are more persistent and tend to be biomagnified up the food chain, the generally greater diversity in plover diet (Smith and Nol 2000; Rose et al. 2016), which includes larger predatory invertebrates, is a possible explanation for the higher concentrations in this species of p,p'-DDE, and also hexa-, hepta-, and octachlorobiphenyls. Overall, the lack of interspecific differences for most contaminants suggests similar contaminant uptake and detoxification processes across species.

Seasonal differences

A key finding in this study was higher concentrations of most organochlorines in the samples from the breeding grounds, implying continued exposure

en route or at the breeding grounds, and suggesting North American rather than Latin American exposure to contaminants at the time of sampling. Changes in most contaminant residue concentrations between collection periods exhibited similar trends. We found seasonal differences for all organochlorine contaminants except Aroclor 1254:1260 and higher chlorinated congeners, with concentrations from the breeding grounds always the highest for chlorinated pesticides. As an individual moves from a contaminated to an uncontaminated site, its contaminant burden should decline and, therefore, lower contaminant concentrations were expected in the Arctic, which is far from pollutant sources. Counter to this prediction, we found higher concentrations of most contaminants in samples collected at the Arctic breeding grounds. One likely possibility for higher organochlorine concentrations in our low Arctic samples is atmospheric transport of pollutants to higher latitudes (Wania and Mackay 1993; Mackay and Wania 1995). The lower chlorinated PCB congeners are more volatile and have a greater tendency to be transported in the atmosphere to higher latitudes (Wania 2003). However, the switch in diet to a greater proportion of freshwater insect larvae during the breeding season may also have an influence (Kidd et al. 1995; Walters et al. 2008). Moreover, remobilization of lipids prior to the initiation of breeding, notably by female birds, could account for the apparent higher levels found at this time of the year; or reduced feeding during breeding could have further enhanced circulation of organochlorines (e.g., Perkins and Barclay 1997, Daley et al. 2014).

Organochlorine concentrations in the spring from Delaware Bay were lower than from the breeding grounds except for DDE, DDD + DDT, dieldrin, and Σ_{45}PCB for which no differences were found. This might suggest contaminant uptake during their short stay on the highly industrialized American east coast. However, stopovers in the spring in Delaware Bay, when birds are en route to the breeding grounds, may be very short, sometimes only a week (Clark et al. 1993; Skagen and Knopf 1994). Thus, concentrations observed on the spring stopover and on the breeding grounds might reflect contaminant uptake in Venezuela or other wintering locations prior to

migration departure in late winter, i.e., staging birds in the spring might not have time to come to equilibrium with the environmental exposure to contaminants. Variation in periods of residency of the birds at the different locations, and tissue turnover time (Hobson and Clark 1992) are expected to play a major role in contaminant uptake. Given the short residency of birds during their northward migration, contaminant levels found from Delaware Bay samples may not reflect contamination at the spring stopover but could have been a carryover effect of exposure at other wintering areas not sampled in this study. Nonetheless, our results suggest that birds migrating south are slowly depurating organochlorine burdens for which levels tend to decrease along the migration journey to Venezuela (e.g., HCB, ΣHCH, HE, ΣCHL, DDE, ΣMirex, Σ_{45}PCB). Samples collected in Venezuela in early winter were likely of birds recently arrived from stopovers in North America and contaminant levels likely reflect uptake from the breeding grounds and the autumn stopover, explaining the general absence of difference in contaminant levels between this site and the breeding grounds and autumn stopover locations. In contrast, late winter birds may have been in the country for 5 months and are likely a better indicator of contaminant exposure at wintering areas.

Similarly, in Venezuela, DDE declined over the winter in sanderling, ruddy turnstone (Arenaria interpres), and least (Charadrius minutilla) and semipalmated sandpipers (Banasch et al. 1992). Terrestrial shorebirds can accumulate very high levels of contaminants as evidenced by organochlorine-induced mortality of long-billed curlew (Numenius americanus) in Oregon (Blus et al. 1985). Our collections were made in coastal lagoons, whereas shorebirds foraging in agricultural areas may have been significantly more exposed. The potential impact of inland sources of pesticides is also suggested by high organochlorine concentrations reported in killdeer (Charadrius vociferus) in the southern USA (DeWeese et al. 1986; Ellis et al. 1989; Hubbard and Schmitt 1989). However, this is unlikely to be important for marine or coastal shorebird species, potentially explaining why, counter to our prediction, contaminant levels were not increased in the southernmost locations.

In Venezuela, plovers, yellowlegs, and sandpipers accumulate large fat reserves during the few weeks following

arrival, rebuilding energy stores; they then lose weight between November and February, and then gain weight prior to the northward migration in May (McNeil and Cadieux 1972; Cramp and Simmons 1983). The body burden of lipophilic compounds is in constant flux depending on the relative rates of ingestion and elimination. Thus, the apparent lower levels of contaminants during lipid catabolism periods (Bustnes 2010; Daley et al. 2014) could be influenced by increased detoxification processes associated with the increased metabolism of migration activities. The reverse is also true in the period of fattening prior to departure from stopovers: increased fat reserves would serve to increase lipid-weight concentrations (Bustnes et al. 2010), and this was somewhat observed between the two pools of lean and fat dowitchers sampled in the Bay of Fundy, i.e., fatter birds appeared to have higher concentration of organochlorines when detected (Appendix Table 4). Absence of clear patterns in contaminant residues among collection sites, including between breeding grounds and autumn stopover and early winter for HCB and PCBs for example, suggests that the physiology of migration in shorebirds cannot solely explain the residue levels we observed. Disparity among individuals with respect to age, sex, body condition, and fat reserves is possible yet could not be tested for due to small sample size or is indistinguishable within our pooled samples.

A common but often challenged assumption of shorebird migration models is that individuals depart when sufficient subcutaneous fat has been deposited. Dunn et al. (1988) found that semipalmated sandpipers refuel for about 15 days in the Bay of Fundy in order to achieve fat loads of 30–40% for the trans-oceanic flight to South America. The timing of collections of plovers, dowitchers, and sandpipers in the Bay of Fundy was consistent with their usual peak numbers, and suggests that few if any birds would have been present for more than 2 weeks prior to being collected (Hicklin 1987). Despite high rates of intake, this is probably too short a time to come to equilibrium with local conditions, i.e., reach maximum levels or, if depurating, minimum levels for a given contaminant. We found a wide range of lipid levels in all species during autumn migration/ staging in the Bay of Fundy suggesting a mix of lean and fat birds, i.e., new arrivals (lean) and those ready to depart (fat). Mean fat levels of sandpipers collected in this study were 15–23% instead of the 30–40% required for the trans-oceanic flight to South America found by Dunn et al. (1988), implying mid-residency, but there may have been considerable individual variation within

pools. Notably, in this study, the two pools of dowitchers grouped by body condition did indeed have very different lipid levels (11.1% and 21.1%), demonstrating that lipid levels are important to consider when assessing contaminants in these long-distant migrants that rely heavily on resources at stopover sites. Nonetheless, this suggests that contaminants found in the Bay of Fundy birds might partly reflect contaminant uptake from the breeding grounds. Similarly, early collection of samples in Delaware Bay, about 3 weeks before normal peak numbers, at least for semipalmated sandpipers (2–5 June; Clark et al. 1993), might have resulted in the lower levels of lipids in samples collected at this spring stopover site, and the absence of patterns in levels of lipids across sites. It is possible that birds did not have time to deposit fat at this site if they had just arrived from wintering areas; thus, contaminant levels observed in spring collected carcasses could reflect contaminant uptakes from the wintering grounds.

Trace elements

Renal and hepatic trace element concentrations found in tissues of shorebirds may be accumulated over broad regions or picked up from point sources such as dredge disposal impoundments (White and Cromartie 1985). Concentrations of trace elements in this study did not differ greatly from those detected in shorebird tissues collected in Europe (Goede 1985; Blomqvist et al. 1987; Ferns and Anderson 1994) or from other areas of North America at that time (Hui 1998). Nonetheless, seasonal variations were found for each trace element considered, while interspecific differences were found only for renal Cd and Se.

Interspecific differences

Interspecific differences in metal levels may reflect differences in diet and microhabitat rather than local contamination as suggested by Burger et al. (2015) who found interspecific differences in feather Hg, Se, and arsenic (As) levels, but also in blood As, Cd, and chromium (Burger et al. 2017) in shorebirds from Delaware Bay. Cd generally correlates with higher trophic position and tissue nitrogen concentration (Øverjordet et al. 2015). Concentrations of renal Cd varied across species with plovers and dowitchers having the highest levels. These patterns suggest that the diets of plovers and dowitchers are more enriched in nitrogen leading to

higher levels of Cd in the two species compared with the others. Surprisingly, however, such patterns did not hold for hepatic Hg concentrations, an element that usually highly correlates with trophic position (Mallory and Braune 2012; Mallory et al. 2015). The differing inter-specific patterns in Cd and in Hg suggest different depurating mechanisms of those elements. Cd uptake rate is much higher than its rate of excretion in animals. As well, Cd accumulates with age, and animals are not able to get rid of their burden very quickly. On the other hand, Hg can be remobilized in different organs depending on the energetic demand and the physiological needs of an individual, which is influenced by the time of the year (e.g., remobilizing into feathers during molt season, into eggs during breeding; Whitney and Cristol 2018). Se concentration was much lower in yellowlegs compared with the three other species suggesting that diet plays a role in Se uptake as has been seen in other species

Seasonal differences

Overall, birds sampled over the winter at the Venezuelan sites had lower hepatic Hg levels than shorebirds from other locations. Molt patterns could be a predominant factor explaining lower hepatic Hg in shorebirds sampled in the winter, but unfortunately no information on plumage was recorded at the time of collection. Mercury is readily transferred to feathers and as much as 90% of the body burden may be eliminated during molt (Braune and Gaskin 1987). It is possible that the birds collected in Venezuela would have already molted their feathers explaining the lower concentrations observed in liver Hg at this site. In a more recent study, Burger et al. (2018) found higher levels of Cd, Hg, and Se in overwintering sandpiper blood samples from Suriname in late winter. The authors suggested accumulation of trace elements in Suriname compared with Delaware Bay, and our results from almost two decades earlier corroborate uptake of trace element on wintering sites for Cd and Se compared with levels found in Delaware Bay. Our findings suggest that Se uptake is likely highest outside of the Arctic and peaks during the non-breeding season, which could imply higher risk of contamination at wintering locations likely resulting from higher anthropogenic influences at those sites. Se input in the environment is associated with irrigation drainage waters, or with naturally rich selenium environments as are found in Venezuela; this could explain this higher prevalence in the wintering samples. For Hg, the higher levels found at higher latitude sites corroborate the general trend observed in Hg levels between higher and lower latitudes in North America, which is usually mirrored in contaminant burdens of various taxa (\Mallory and Braune 2012; Perkins et al. 2016; Mallory et al. 2017).

Conclusion

Shorebirds are vulnerable to a variety of organic pollutants and trace elements. We found that both regional and interspecific differences occurred across the suite of elements and organochlorine contaminants examined in Nearctic shorebird tissues over their migratory cycle. There was no indication that the shorebird species examined acquired higher contaminants in the southern, non-breeding regions as predicted from continuing pesticide use or contaminant sources. Nor did we find that foraging niche could solely explain levels of contaminants, although stable isotope analysis would have enabled more detailed insights on this aspect. Collectively, the explanation for patterns of contamination in shorebirds is more complex than can be determined by geographic location or foraging niche, and continental scale exposure as well as metabolism of contaminants are likely to play a role in the levels of contaminants detected in long-range migrant shorebirds.

Appendix

Table 4 Concentrations ($\mu g\ g^{-1}$ lipid wt) of major organochlorine contaminants in carcasses of four migrant shorebird species at different stages of their migratory cycle

Species	Season	Age	Sex	Number	% lipids	HCB	ΣHCH	Oxychlordane	ΣCHL	HE	p,p'-DDE	p,p'-DDT + p,p'-DDD	ΣMirex	Dieldrin	ΣPCB	Aroclor 1254:1260
Lesser yellowlegs	Breeding	A	M	5	9.7	0.031	0.021	0.062	0.240	0.082	2.557	Trace	0.052	0.072	0.598	1.186
		A	F	5	5.2	ND	0.019	Trace	0.027	Trace	29.154	0.058	0.058	0.038	0.731	ND
	Autumn migration	A	U	1	1.4	0.071	ND	Trace	0.043	ND	12.143	ND	Trace	ND	0.786	1.143
		HY	U	7	16.1	0.019	Trace	ND	0.035	0.019	0.211	ND	0.031	0.019	1.062	1.814
	Early winter	A	M/F	7	14.6	ND	ND	0.055	0.073	0.021	3.068	ND	0.048	0.041	0.370	1.274
		HY	U	3	4.8	0.021	ND	Trace	0.017	ND	0.604	ND	0.063	ND	1.938	0.958
	Late winter	A	M	7	14.7	Trace	ND	Trace	0.001	ND	0.762	ND	0.027	0.007	0.306	0.558
		A	F	3	12.3	Trace	ND	Trace	0.001	ND	0.398	ND	0.016	ND	0.366	0.699
	Spring migration	A	M	1	27.0	Trace	ND	0.011	0.040	0.007	0.559	0.022	0.004	1.352	0.400	0.522
Semipalmated plover	Breeding	A	M	5	12.4	0.056	0.040	0.137	0.361	0.081	8.177	0.056	0.661	2.073	3.403	5.371
		A	F	5	9.2	0.043	0.033	0.120	0.405	0.152	1.543	0.033	0.348	1.163	1.554	2.272
	Autumn migration	A	M	5	19.4	0.036	0.057	0.191	0.314	ND	7.082	0.031	0.015	ND	1.619	2.861
		A	F	4	33.7	0.015	ND	0.021	0.037	ND	0.475	ND	0.018	0.024	0.534	1.039
	Early winter	A	M/F	6	12.8	0.070	0.016	0.078	0.381	0.055	7.172	3.813	0.086	0.203	3.984	5.367
		FY	M	3	12.0	Trace	ND	ND	ND	ND	0.175	ND	0.025	0.033	2.675	1.183
		HY	F	1	9.1	0.044	ND	0.033	0.077	0.022	0.198	ND	Trace	0.066	0.286	0.341
	Late winter	A	M	6[a]	10.4	Trace	ND	0.010	0.012	ND	2.260	ND	0.010	Trace	0.788	1.471
		A	F	4	12.4	Trace	ND	0.024	0.026	Trace	0.161	ND	0.008	Trace	0.524	0.871
	Spring migration	A	M	5	14.7	0.007	ND	0.020	0.084	0.007	3.741	0.102	0.027	0.088	2.061	3.980
		A	F	5	13.8	0.007	0.029	0.043	0.179	0.036	3.217	0.333	0.036	0.188	2.022	3.275
Semipalmated sandpiper	Breeding	A	M	5	13.7	0.073	0.015	0.044	0.066	0.044	0.350	0.051	0.058	0.029	1.839	3.058
		A	F	5	11.8	0.042	0.008	0.025	0.124	0.017	0.246	Trace	0.025	0.017	0.890	1.703
	Autumn migration	A	M	5	15.3	0.065	0.020	0.085	0.193	0.052	0.085	0.013	0.098	0.039	0.359	0.869
		A	F	5	22.8	0.022	ND	0.031	0.080	0.018	0.075	Trace	0.026	0.022	1.175	2.873
	Early winter	A	M/F	7	20.3	0.020	ND	0.074	0.198	0.010	0.118	0.015	0.039	0.025	1.438	2.700
		FY	M	1	18.9	0.005	ND	0.032	0.046	0.011	0.037	ND	0.063	0.021	1.344	2.704
		HY	M/F	9	12.9	0.047	ND	0.047	0.107	0.023	0.070	ND	0.016	0.047	0.248	0.372
	Late winter	A	M	9[a]	14.6	Trace	ND	0.055	0.053	Trace	0.021	ND	0.007	0.034	0.432	0.877
		A	F	4[a]	12.2	Trace	ND	ND	ND	ND	0.016	ND	Trace	ND	0.172	0.336

Table 4 (continued)

Species	Season	Age	Sex	Number	% lipids	HCB	ΣHCH	Oxychlordane	ΣCHL	HE	p,p'-DDE	p,p'-DDT + p,p'-DDD	ΣMirex	Dieldrin	ΣPCB	Aroclor 1254:1260
	Spring	A	M	5	14.6	0.014	ND	0.014	0.055	0.014	0.452	0.110	0.014	0.027	0.534	0.788
	migration	A	F	5	14.1	0.007	ND	ND	0.023	0.014	0.674	Trace	0.028	0.078	0.957	1.957
Short-billed dowitcher	Breeding	A	M	5	7.3	0.041	Trace	0.384	2.479	0.233	3.575	0.096	0.370	0.452	3.370	5.178
		A	F	5	6.0	0.017	ND	0.033	0.280	0.033	4.367	0.017	0.533	0.167	0.933	ND
	Autumn	A	M	4	21.1	ND	ND	ND	ND	ND	0.460	ND	ND	ND	0.682	1.507
	migration	A	M	4	11.1	ND	ND	ND	0.088	ND	0.883	ND	0.063	0.054	3.892	9.054
	Early winter	A	M/F	8	13.9	0.014	0.007	0.029	0.138	0.014	1.029	0.029	0.029	0.065	1.842	3.719
		FY	F	2	12.7	Trace	ND	Trace	0.040	ND	0.362	0.008	0.047	0.039	0.890	1.488
	Late winter	A	M	8[b]	17.2	Trace	ND	Trace	0.008	ND	0.390	Trace	0.006	0.006	0.448	0.988
		A	F	3	15.0	Trace	ND	ND	0.003	ND	0.167	ND	Trace	Trace	0.353	0.540
	Spring	A	M	4	19.1	0.005	ND	ND	0.023	0.010	1.356	0.037	0.016	0.021	1.063	1.419
	migration	A	F	7	16.9	0.006	0.006	0.018	0.098	0.012	1.018	0.189	0.012	0.047	0.828	1.272

Values are for pools of (N) samples μg g^{-1} on a lipid wt basis. *M* male, *F* female, *U* unrecorded, *M/F* both sexes

HCB hexachlorobenzene, *ΣHCH* sum of α-, β-, and γ-hexacyclohexane, *HE* heptachlor epoxide

Aroclor 1254:1260 = PCB congener 138 basis

ND non-detected, trace = values <0.001 wet wt

[a] Includes one first-year bird

[b] Includes two first-year birds

Table 5 Concentrations of metals (μg g^{-1} dry wt) in shorebird tissues at different stages of their migratory cycle

Species	Season	Age	Sex	N	Kidney %H$_2$O	Cd	Se	Liver %H$_2$O	Hg
Lesser yellowlegs	Breeding season	A	F	5	72		4.80	69	2.69
	Autumn migration	A	M	5	73	1.50	2.82	68	2.56
		J	U	7	68	0.60	3.59	72	2.40
		A	U	1	71	0.77	4.28	69	1.75
	Early winter	HY	F/M	3	70	Trace	4.72	68	0.78
		A	F/M	7	73	3.85	7.61	68	0.32
	Late winter	A	F	3	67	1.08	5.41	70	0.34
		A	M	7	72	3.80	5.10	66	1.72
	Spring migration	A	M	1	67	3.22	6.29	66	2.03
Semipalmated plover	Breeding season	A	M	5	70	5.97	3.33	66	1.08
	Autumn migration	A	F	5	n/a	3.28		67	
		A	M	5	66	Trace	6.11	67	1.01
	Early winter	A	F	4	71	8.33	7.82	67	0.41
	Early winter	HY	F	1	69	12.7	16.4	65	1.29
		FY	M	3	68	6.5	9.38	70	1.36
	Late winter	A	F/M	6[a]	69	10.3	9.01	72	0.58
	Spring migration	A	M	4	71	6.2	13.6	69	1.75
		A	F	5	71	3.5	6.71	69	1.61
Semipalmated sandpiper	Breeding season	A	F/M	5	56	2.28	7.02	67	1.37
		A	M	5	70		7.38		
	Autumn migration	A	F	5	66	1.87	6.98	64	2.13
	Early winter	HY	F/M	9	67	0.85	7.81	69	2.98
		A	F/M	7	66	4.11	6.72	66	0.96
		FY	M	1	66	9.28		66	0.26
	Late winter	A	M	9[a]	66	4.37	8.17	71	0.46
		A	F	4[a]	69	3.57	15.8	66	0.22
	Spring migration	A	F	5	69	1.90	9.49	68	0.89

Table 5 (continued)

Species	Season	Age	Sex	N	Kidney %H$_2$O	Cd	Se	Liver %H$_2$O	Hg
Short-billed dowitcher	Breeding season	A	M	5	69	2.54	6.41	67	0.97
		A	F	5	n/a	7.35		66	1.60
	Autumn migration	A	M	5	70		5.86		
		A	M	4	71	3.96	5.51	58	1.62
		A	M	4	67	4.69	7.60	70	0.12
	Early winter	FY	F	2	71	4.21	8.04	69	0.43
		A	F/M	8	71	4.53	14.1	67	0.58
	Late winter	A	M	8[b]	69	7.33	11.0	69	0.17
		A	F	3	69	2.18	9.06	70	1.53
	Spring migration	A	M	4	70	1.89	12.4	75	2.44
		A	F	7	72				

ND non-detected

[a] Includes one first-year bird

[b] Includes two first-year bird

References

Amano, T., Szekely, T., Koyama, K., Amano, H., & Sutherland, W. J. (2010). A framework for monitoring the status of populations: an example from wader populations in the East Asian-Australasian flyway. *Biological Conservation, 143*, 2238–2247. https://doi.org/10.1016/j.biocon.2010.06.010.

Andres, B. A., Smith, P. A., Morrison, R. I. G., Gratto-Trevor, C. L., Brown, S. C., & Friis, C. A. (2012). Population estimates of North American shorebirds, 2012. *Wader Study Group Bulletin, 119*, 178–192.

Ballschmiter, K., & Zell, M. (1980). Analysis of polychlorinated biphenyls (PCB) by glass capillary gas chromatography. . Composition of technical Arochlor- and Clophen-PCB mixtures. *Fresenius' Zeitschrift für analytische Chemie, 302*, 20–31.

Banasch, U., Goossen, J. P., Riez, A. E., Casler, C., & Barradas, R. D. (1992). Organochlorine contaminants in migrant and resident prey of peregrine falcons, *Falco peregrinus*, in Panama, Venezuela and Mexico. *Canadian Field-Naturalist, 106*, 493–498.

Bart, J., Brown, S., Harrington, B., & Morrison, R. I. G. (2007). Survey trends of North American shorebirds: population declines or shifting distributions? *Journal of Avian Biology, 38*, 73–82 https://10.1111/j.2007.0908-8857.03698.x.

Beyer, W. N., & Meador, J. P. (2011). *Environmental contaminants in biota: interpreting tissue concentrations*. Boca Raton: CRC Press, Taylor and Francis Group.

Blomqvist, S., Frank, A., & Petersson, L. R. (1987). Metals in liver and kidney tissues of autumn-migrating dunlin *Calidris alpina* and curlew sandpiper *Calidris ferruginea* staging at the Baltic Sea. *Marine Ecology Progress Series, 35*, 1–13.

Blus, L. J., Henny, C. J., & Krynitsky, A. (1985). Organochlorine-induced mortality and residues in long-billed curlews from Oregon. *Condor, 87*, 563–565.

Braune, B. M., & Gaskin, D. E. (1987). Mercury levels in Bonaparte's gulls (*Larus philadelphia*) during autumn molt in the Quoddy region, New Brunswick, Canada. *Archives of Environmental Contamination and Toxicology, 16*, 539–549.

Braune, B. M., & Mallory, M. L. (2017). Declining trends of polychlorinated dibenzo-p-dioxins, dibenzofurans and non-ortho PCBs in Canadian Arctic seabirds. *Environmental Pollution, 220*, 557–566 https://10.1016/j.envpol.2016.10.003.

Braune, B. M., & Noble, D. G. (2009). Environmental contaminants in Canadian shorebirds. *Environmental Monitoring and Assessment, 148*, 185–204 https://10.1007/A10661-007-0150-0.

Braune, B. M., Gaston, A. J., & Mallory, M. L. (2018). Temporal trends of legacy organochlorines in eggs of Canadian Arctic seabirds monitored over four decades. *Science of the Total Environment, 646*, 551–563 https://10.1016/j.scitotenv.2018.07.291.

Burger, J., Larry, N., & Clark, K. E. (1992). Importance of beach, mudflat and marsh habitats to migrant shorbirds on Delaware Bay. *Biological Conservation, 79*, 1202–1205.

Burger, J., Tsipoura, M., Niles, L. J., Gochfeld, M., Dey, A., & Mizrahi, D. (2015). Mercury, lead, cadmium, arsenic, chromium and selenium in feathers of shorebirds during migrating through Delaware Bay, New Jersey: comparing the 1990s and 2011/2012. *Toxics, 3*, 63–74.

Burger, J., Tsipoura, N., & Gochfeld, M. (2017). Metal levels in blood of three species of shorebirds during stopover on Delaware Bay reflect levels in their food, horseshoe crab eggs. *Toxics*. https://doi.org/10.3390/toxics5030020.

Burger, J., Mizrahi, D., Tsipoura, N., Jeitner, C., & Gochfeld, M. (2018). Mercury, lead, cadmium, cobalt, arsenic and selenium in the blood of semipalmated sandpipers (*Calidris pusilla*) from Suriname, South America: age-related differences in wintering site and comparisons with stopover site in New Jersey, USA. *Toxics, 6*, 1–11. https://doi.org/10.3390/toxics6020027.

Bustnes, J. O., Moe, B., Herzke, D., Hanssen, S. A., Nordstad, T., Sagerup, K., Gabrielsen, G. W., & Borgå, K. (2010). Strongly increasing blood concentrations of lipid-soluble organochlorines in high arctic common eiders during incubation fast. *Chemosphere, 79*, 320–325.

Clark, K. E., Niles, L. J., & Burger, J. (1993). Abundance and distribution of migrant shorebirds in Delaware Bay. *Condor, 95*, 694–705.

Cramp, S., & Simmons, K. E. L. (1983). *The birds of the western Palearctic* (Vol. III). Oxford: Oxford university Press.

Daley, J. M., Paterson, G., & Drouillard, K. G. (2014). Bioamplification as a bioaccumulation mechanism for persistent organic pollutants (POPs) in wildlife. *Reviews of Environmental Contamination and Toxicology, 227*, 107–155.

DeWeese, L. R., McEwen, L. C., Hensler, G. L., & Petersen, B. E. (1986). Organochlorine contaminants in passeriformes and other avian prey of the peregrine falcon in the western United States. *Environmental Toxicology and Chemistry, 5*, 675–693.

Dias, R. A., Blanco, D. E., Goijman, A. P., & Zaccagnini, M. E. (2014). Density, habitat use, and opportunities for conservation of shorebirds in rice fields in southeastern South America. *Condor, 116*, 384–393. https://doi.org/10.1650/CONDOR-13-160.1.

Didyk, A. S., & Burt, M. D. B. (1997). *Himasthla limnodromi* n. sp. (Digenea: Echinostomatidae) from the short-billed dowitcher, *Limnodromus griseus* (Aves: Scolopacidae). *Journal of Parasitology, 83*, 1124–1127.

Didyk, A. S., & Burt, M. D. B. (1998a). *Paraliga charadrii* n. sp. (Cestoda: Dilepididae) from the semipalmated plover, *Charadrius semipalmatus* (Aves: Charadriiformes). *Journal of Parasitology, 84*, 828–830.

Didyk, A. S., & Burt, M. D. B. (1998b). Geographical, seasonal and sex dynamics of *Shipleya inermis* (Cestoidea: Dioecocestidae) in *Limnodromus griseus* (Gemelin (Aves: Charadriiformes)). *Journal of Parasitology, 84*, 931–934.

Dunn, P. O., May, T. A., & McCollough, M. A. (1988). Length of stay and fat content of migrant semipalmated sandpipers in eastern Maine. *Condor, 90*, 824–835.

Ellis, D. H., DeWeese, L. R., Grubb, T. G., Kiff, L. F., Smith, D. G., Jarman, W. M., & Peakall, D. B. (1989). Pesticide residues in Arizona peregrine falcon eggs and prey. *Bulletin of Environmental Contamination and Toxicology, 42*, 57–64.

Ferns, P. N., & Anderson, J. I. (1994). Cadmium in the diet and body-tissues of dunlin *Calidris alpina*, from the Bristol Channel, UK. *Environmental Pollution, 86*, 225–231.

Flemming, S. A., Nol, E., Kennedy, L. V., & Smith, P. A. (2019). Hyperabundant herbivores limit habitat availability and influence nest site selection of Arctic-breeding birds. *Journal of Applied Ecology, 56*, 976–998. https://doi.org/10.1111/1365-2664.13336.

Furness, R. W., & Camphuysen, K. (1997). Seabirds as monitors of the marine environment. *ICES Journal of Marine Science, 54*, 726–737.

Gerwing, T. G., Kim, J.-H., Hamilton, D. J., Barbeau, M. A., & Addison, J. A. (2016). Diet reconstruction using next-generation sequencing increases the known ecosystem usage by a shorebird. *Auk, 133*, 168–177. https://doi.org/10.1642/auk-15-176.1.

Goede, A. A. (1985). Mercury, selenium, arsenic and zinc in waders from the Dutch Wadden Sea. *Environmental Pollution, 37*, 287–309.

Gratto-Trevor, C., Morrison, R. I. G., Collins, B., Rausch, J., Drever, M., & Johnston, V. (2011). *Trends in Canadian shorebirds. Canadian Biodiversity: Ecosystem Status and Trends 2010*. Ottawa: Environment Canada.

Hargreaves, A. L., Whiteside, D. P., & Gilchrist, G. (2010). Concentrations of 17 elements, including mercury, and their relationship to fitness measures in arctic shorebirds and their eggs. *Science of the Total Environment, 408*, 3153–3161.

Hicklin, P. W. (1987). The migration of shorebirds in the bay of Fundy. *Wilson Bulletin, 99*, 540–570.

Hicklin, P., & Gratto-Trevor, C. L. (2010). Semipalmated sandpiper (*Calidris pusilla*), version 2.0. In A. F. Poole (Ed.), *The birds of North America online*. Ithaca: Cornell Lab of Ornithology. https://doi.org/10.2173/bna.6 Accessed 18 June 2019.

Hobson, K. A., & Clark, R. G. (1992). Assessing avian diets using stable isotopes II: factors influencing diet-tissue fractionation. *Condor, 94*, 189–197.

Howe, M. A., Giessler, P. H., & Harrington, B. A. (1989). Population trends of North American shorebirds based on the International Shorebird Survey. *Biological Conservation, 49*, 185–199.

Hubbard, J. P., & Schmitt, C. G. (1989). Organochlorine residues in avian prey of peregrine falcons breeding in New Mexico. *National Wildlife Federal Science Technical Service, 11*, 176–181.

Hui, C. A. (1998). Metal and trace element burdens in two shorebird species at two sympatric wintering sites in southern California. *Environmental Monitoring and Assessment, 50*, 233–247.

Hui, C. A., & Beyer, W. N. (1998). Sediment ingestion of two sympatric shorebird species. *Science of the Total Environment, 224*, 227–233.

Jayaraj, R., Megha, P., & Sreedev, P. (2016). Organochlorine pesticides, their toxic effects on living organisms and their fate in the environment. *Interdisciplinary Toxicology, 9*, 90–100 https://10.1515/intox-2016-0012.

Jehl Jr., J. R., Klima, J., & Harris, R. E. (2001). Short-billed Dowitcher (*Limnodromus griseus*), *version 2.0*. In A. F. Poole (Ed.), *The birds of North America online*. Ithaca: Cornell Lab of Ornithology. https://doi.org/10.2173/bna.564 Accessed 18 June 2019.

Johnstone, R. M., Court, G. S., Fesser, A. C., Bradley, D. M., Oliphant, L. W., & McNeil, J. D. (1996). Long-term trends and sources of organochlorine contamination in Canadian

tundra peregrine falcons, *Falco peregrinus tundrius*. *Environmental Pollution, 93*, 109–120.

Kidd, K. A., Schindler, D. W., Hesslein, R. H., & Muir, D. C. G. (1995). Correlation between stable nitrogen isotope ratios and concentrations of organochlorines in biota from a freshwater food web. *Science of the Total Environment, 160*, 381–390.

Lamarre, J.-F., Legagneux, P., Gauthier, G., Reed, E. T., & Bêty, J. (2017). Predator-mediated negative effects of overabundant snow geese on arctic-nesting shorebirds. *Ecosphere, 8*, e01788. https://doi.org/10.1002/ecA2.1788.

Letcher, R. J., Bustnes, J. O., Dietz, R., Jenssen, B. M., Jorgensen, E. H., Sonne, C., Verreault, J., Vijayan, M. M., & Gabrielsen, G. W. (2010). Exposure and effects assessment of persistent organohalogen contaminants in arctic wildlife and fish. *Science of the Total Environment, 408*, 2995–3043.

Loganathan, B. G., Kannan, K. (1994). Global organochlorine contaminantion trends: an overview. *AMBIO A Journal of the Human Environment*, 3–31.

MacDonald, E. C., Ginn, M. G., & Hamilton, D. J. (2012). Variability in foraging behavior and implications for diet breadth among semipalmated sandpipers staging in the Upper Bay of Fundy. *Condor, 114*, 135–144. https://doi.org/10.1525/cond.2012.100246.

Mackay, D., & Wania, F. (1995). Transport of contaminants to the Arctic: partitioning, processes and models. *Science of the Total Environment, 160*, 25–38.

Mallory, M. L., & Braune, B. M. (2012). Tracking contaminants in seabirds of Arctic Canada: temporal and spatial insights. *Marine Pollution Bulletin, 64*, 1475–1484.

Mallory, M. L., Braune, B. M., Provencher, J. F., Callaghan, D. B., Gilchrist, H. G., Edmonds, S. T., Allard, K., & O'Driscoll, N. J. (2015). Mercury levels in feathers of marine birds in Arctic Canada. *Marine Pollution Bulletin, 98*, 308–313.

Mallory, C. D., Gilchrist, H. G., Robertson, G. J., Provencher, J. F., Braune, B. M., Forbes, M. R., & Mallory, M. L. (2017). Hepatic trace element concentrations of breeding female common eiders across a latitudinal gradient in the eastern Canadian Arctic. *Marine Pollution Bulletin, 124*, 252–257.

McCloskey, M., Robinson, S., Smith, P. A., & Forbes, M. (2013). Mercury concentration in the eggs of four Canadian Arctic-breeding shorebirds not predicted based on their population statuses. *Springerplus, 2*, 567. https://doi.org/10.1186/2193-1801-2-567.

McNeil, R., & Cadieux, F. (1972). Fat content and flight range capabilities of some adult spring and fall migrant North American shorebirds in relation to migration routes on the Atlantic coast. *Naturaliste Canadien, 99*, 589–606.

Morrison, R. I. G., Bourget, A., Butler, R., Dickson, H.L., Gratto-Trevor, C., Hicklin, P., Hyslop, C., Ross, R. K. (1994). *A preliminary assessment of the status of shorebird populations in Canada*. Canadian Wildlife Service Program Notes No. 208. Ottawa. 19 p.

Morrison, R. I. G., Mizrahi, D. S., Ross, R. K., Ottema, O. H., de Pracontal, N., & Narine, A. (2012). Dramatic declines of semipalmated sandpipers on their major wintering areas in the Guianas, Northern South America. *Waterbirds, 35*, 120–134.

Myers, J. P. (1986). Sex and gluttony on Delaware Bay. *Natural History, 95*, 68–77.

Neugebauer, E. A., Sans Cartier, G. L., Wakeford, B. J. (2000). *Methods for the determination of metals in wildlife tissues using various atomic absorption spectrophotometry techniques*. (Technical Report Series Number 337). Hull, Canada: Canadian Wildlife Service.

Nicholls, C. I., & Altieri, M. A. (1995). Conventional agricultural development models and the persistence of the pesticide treadmill in Latin America. *International Journal of Sustainable Development and World Ecology, 4*, 93–111.

Nol, E., & Blanken, M. S. (2014). Semipalmated plover (Charadrius semipalmatus), version 2.0. In A. F. Poole (Ed.), *The birds of North America online*. Ithaca: Cornell Lab of Ornithology. https://doi.org/10.2173/bna.444 Accessed 18 June 2019.

Norstrom, R. J., & Won, H. T. (1985). Long-term preservation of egg and tissue homogenates for determination of organochlorine compounds: freezing versus freeze-drying. *Journal-Association of Official Analytical Chemists, 68*, 129–135.

North American Bird Conservation Initiative Canada. (2019). *The state of Canada's birds, 2019*. Ottawa: Environment and Climate Change Canada https://www.stateofcanadabirds.org.

O'Reilly, K. M., & Wingfield, J. C. (2003). Seasonal, age, and sex differences in weight, fat reserves, and plasma corticosterone in western sandpipers. *Condor, 105*, 13–26.

Øverjordet, I. B., Gabrielsen, G. W., Berg, T., Ruus, A., Evenset, A., Borgå, K., Christensen, G., Lierhagen, S., & Jenssen, B. M. (2015). Effect of diet, location and sampling year on bioaccumulation of mercury, selenium and cadmium in pelagic feeding seabirds in Svalbard. *Chemosphere, 122*, 14–22.

Page, G. W., Stenzel, L. E., & Kjelmyr, J. E. (1999). Overview of shorebird abundance and distribution in wetlands of the Pacific coast of the contiguous United States. *Condor, 101*, 461–471.

Pearce-Higgins, J. W., Brown, D. J., Douglas, D. J. T., Alves, J. A., Bellio, M., Bocher, P., Buchanan, G. M., Clay, R. P., Conklin, J., et al. (2017). A global threats overview for Numeniini populations: synthesising expert knowledge for a group of declining migratory birds. *Bird Conservation International, 27*, 6–24 https://10.1017/S0959270916000678.

Perkins, C. R., & Barclay, J. S. (1997). Accumulation and mobilization of organochlorine contaminants in wintering greater scaup. *The Journal of Wildlife Management, 61*, 444–449.

Perkins, M., Ferguson, L., Lanctot, R. B., Stenhouse, I. J., Kendall, S., Brown, S., Gates, H. R., Hall, J. O., Regan, K., & Evers, D. C. (2016). Mercury exposure and risk in breeding and staging Alaskan shorebirds. *Condor, 118*, 571–581.

Petit, M., Vézina, F., & Piersma, T. (2010). Ambient temperature does not affect fuelling rate in absence of digestive constraints in long-distance migrant shorebird fuelling up in captivity. *Journal of Comparative Physiology, 180*, 847–856.

Pfister, C., & Kasprzyk, M. J. (1998). Body-fat levels and annual return in migrating semipalmated sandpipers. *Auk, 115*, 904–915.

Piersma, T., & van Gils, J. A. (2011). *The flexible phenotype: a body-centred integration of ecology, physiology, and behaviour*. London: Oxford University Press.

Quinn, J. T., & Hamilton, D. J. (2012). Variation in diet of semipalmated sandpipers (Calidris pusilla) during stopover in the upper bay of Fundy, Canada. *Canadian Journal of Zoology, 90*, 1181–1190 https://10.1139/Z2012-086.

R Core Team (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Roberts, D. R., Laughlin, L. L., Hsheih, P., & Legters, L. J. (1997). DDT, global strategies, and a malaria control crisis in South America. *Emerging Infectious Dieseases, 3*, 295–302.

Rose, M., Pollock, L., & Nol, E. (2016). Diet and prey size selectivity of semipalmated plovers in coastal Georgia. *Canadian Journal of Zoology, 94*, 727–732.

Saalfeld, D. T., Matz, A. C., McCaffery, B. J., Johnson, O. W., Bruner, P., & Lanctot, R. B. (2016). Inorganic and organic contaminants in Alaskan shorebird eggs. *Environmental Monitoring and Assessment, 188*, 276. https://doi.org/10.1007/s10661-016-5270-y.

Skagen, S. K., & Knopf, F. L. (1994). Residency patterns of migrating sandpipers at a midcontinental stopover. *Condor, 96*, 949–958.

Skagen, S. K., & Oman, H. D. (1996). Dietary flexibility of shorebirds in the western hemisphere. *Canadian Field-Naturalist, 110*, 419–444.

Smith, A. C., & Nol, E. (2000). Winter foraging behaviour and prey selection of the semipalmated plover in coastal Venezuela. *Wilson Bulletin, 112*, 467–472.

Smith, R. V., Stafford, J. D., Yetter, A. P., Horath, M. M., & Hine, C. S. (2012). Foraging ecology of fall-migrating shorebirds in the Illinois River valley. *PLoS One, 7*, 45121. https://doi.org/10.1371/journal.pone.0045121.

Sutherland, W. J., Alves, J. A., Amano, T., Chang, C. H., Davidson, N. C., Finlayson, C. M., Gill, J. A., et al. (2012). A horizon scanning assessment of current and potential future threats to migratory shorebirds. *Ibis, 154*, 663–679.

Tanabe, S. (2002). Contamination and toxic effects of persistent endocrine disrupters in marine mammals and birds. *Marine Pollution Bulletin, 45*, 69–77.

Tanaka, H., Ogi, H., Tanabe, S., Tatsukawa, R., & Oka, N. (1986). Bioaccumulation and metabolism of PCBs and DDE in short-tailed shearwater *Puffinus tenuirostris* during its transequatorial migration and in the wintering grounds. *Memoirs of National Institute of Polar Research. Special issue, 40*, 434–442.

Tibbits, T. L., & Moskoff, W. (2014). Lesser yellowlegs (*Tringa flavipes*), version 2.0. In A. F. Poole (Ed.), *The birds of North America online*. Ithaca: Cornell Lab of Ornithology. https://doi.org/10.2173/bna.427 Accessed 18 June 2019.

Tsipoura, N., Burger, J., Niles, L., Dey, A., Gochfeld, M., Peck, M., & Mizrahi, D. (2017). Metal levels in shorebirds feathers and blood during migration through Delaware Bay. *Archives of Environmental Contamination and Toxicology, 72*, 562–574.

Turle, R., & Collins, B. (1992). Validation of the use of pooled samples for monitoring of contaminants in wildlife. *Chemosphere, 25*, 463–469.

van den Brink, N. W., van Franeker, J. A., & de Ruiter-Dijkman, E. M. (1998). Fluctuating concentrations of organochlorine pollutants during a breeding season in two Antarctic seabirds: Adélie penguin and southern fulmar. *Environmental Toxicology and Chemistry, 17*, 702–709.

Wakeford, B., & Turle, R. (1997). In-house reference materials as a means to quality assurance: the Canadian Wildlife Service experience. In R. E. Clement: L. H. Keith, & K. W. M. Siu

(Eds.), *Reference materials for environmental analysis* (pp. 205–231). Boca Raton: CRC Press.

Walters, D. M., Fritz, K. M., & Otter, R. R. (2008). The dark side of subsidies: adult stream insects export organic contaminants to riparian predators. *Ecological Applications, 18*, 1835–1841.

Wania, F. (2003). Assessing the potential of persistent organic chemicals for long-range transport and accumulation in polar regions. *Environmental Science and Technology, 37*, 1344–1351.

Wania, F., & Mackay, D. (1993). Global fractionation and cold condensation of low volatility organochlorine compounds in polar regions. *Ambio, 22*, 10–18.

Watts, B. D., Reed, E. T., & Turin, C. (2015). Estimating sustainable mortality limits for shorebirds using the Western Atlantic Flyway. *Wader Study, 122*, 37–53.

White, D. H., & Cromartie, E. (1985). Bird use and heavy metal accumulation in waterbirds at dredge disposal impoundments, Corpus Christi, Texas. *Bulletin of Environmental Contamination and Toxicology, 34*, 295–300.

Whitney, M. C., & Cristol, D. A. (2018). Impacts of sublethal mercury exposure on birds: a detailed review. *Reviews of Environmental Contamination and Toxicology, 244*, 113–163.

Young, D. R., Gossett, R. W., & Heesen, T. C. (1988). Persistence of chlorinated hydrocarbon contamination in a California marine ecosystem. In D. A. Wolfe & T. P. O'Connor (Eds.), *Oceanic processes in marine pollution. Vol. 5. Urban wastes in coastal marine environments* (pp. 33–41). FL: Krieger Malabar.