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Abstract

Chronic exposure to pollutants may represent a threat for wildlife. We tested whether adult survival rate, breeding probability and breeding success the year of sampling and the following year were affected by blood levels of mercury or persistent organic pollutants in Svalbard black-legged kittiwake *Rissa tridactyla*, by using capture–mark–recapture models over a five-year period. Survival rate was negatively linked to HCB levels in females, to chlordane mixture and oxychlordane, tended to decrease with increasing PCBs or DDE levels, but was unrelated to mercury. Breeding probability decreased with increasing mercury levels during the sampling year and with increasing CHL or HCB levels during the following year, especially in males observed as breeders. Surprisingly, the probability of raising two chicks increased with increasing HCB levels. Although levels of these legacy pollutants are expected to decline, they represent a potential threat for adult survival rate and breeding probability, possibly affecting kittiwake population dynamics.

Capsule abstract: Negative effects of pollutants were detected on future breeding probabilities and on adult survival rate in a High Arctic seabird species.

Keywords: heavy metals, kittiwake, population, pesticides, PCBs

1. Introduction

Contaminants, such as mercury (Hg) and persistent organic pollutants (hereafter POPs) may represent a threat for wildlife, because of their detrimental effects on developmental, neurological, physiological, endocrine and immune functions (Barron et al., 1995; Bustnes et al., 2003a; Tan et al., 2009; Letcher et al., 2010). Despite a growing environmental concern during the last decades, the demographic consequences of pollution remain poorly evaluated in free-living vertebrates. Only a few long-term monitoring studies have addressed the consequences of environmental pollutants on survival rate and long-term reproductive outputs. Hg or POP levels were negatively related to long-term breeding probability and success in the wandering albatross *Diomedea exulans* and in two *Catharacta skua* species (Goutte et al., 2014a,b). Apparent survival rate was lower in glaucous gulls *Larus hyperboreus*, bearing the highest levels of oxychlordane, a metabolite of the chlordane mixture, which is regarded as one of the most toxic POPs (Erikstad et al., 2013). However, adult survival rate was not related to POPs or Hg in tree swallows (*Tachycineta bicolor*), king eiders (*Somateria spectabilis*), white-winged scoters (*Melanitta fusca*), wandering albatrosses and two *Catharacta* skua species (Wayland et al., 2008; Hallinger et al., 2011; Goutte et al. 2014a,b).

Some seabird species appear as ideal models for assessing the demographic consequences of environmental pollution. Firstly, individual detection probabilities of seabirds at breeding colonies are generally high because of high overall site fidelity (e.g. Gauthier et al., 2012). Secondly, large sample sizes and accurate measures of breeding outputs are relatively easy to obtain in seabird’s colonies. Thirdly, these long-lived top predators are particularly exposed to contaminants, because of bioaccumulation process and biomagnification along the trophic web (Rowe, 2008; Letcher et al., 2010).

The present study focusses on black-legged kittiwakes *Rissa tridactyla* breeding in Svalbard, a Norwegian archipelago in the north-western part of the Barents Sea. The Norwegian Arctic is recognized as a final sink for organic and metallic pollutants, which are transported by atmospheric and oceanic currents and by large rivers (Gabrielsen and Henriksen, 2001). Previous studies in this population of Svalbard kittiwakes have reported deleterious effects of Hg and POPs on endocrine mechanisms (Nordstad et al., 2012; Tartu et al., 2013, 2014). The estimated number of breeding pairs in the Svalbard archipelago is 270 000 in 215 colonies (Strøm, 2006). The status of black-legged kittiwakes is near threatened, with a pronounced population decline from 1995 to 2002 and a slight increase from 2002 to 2012 (Barrett et al., 2012). This study aims at detecting whether breeding probability the year of sampling and demographic traits the following year (apparent adult survival rate, breeding probability, probability of successfully raising at least one chick and probability of successfully raising two chicks) were correlated with individual blood levels of Hg or POPs. According to the few available long-term studies on polar seabird species (Erikstad et al., 2013; Goutte et al., 2014a,b), we predicted deleterious effects of Hg or POPs on breeding probability and breeding success during the year of sampling and during the following year and deleterious effects of the chlordane mixture and metabolites on survival rate in black legged kittiwakes.

2. Materials and methods

2.1. Study area and birds

Our study was conducted in a colony of black legged kittiwakes at Kongsfjorden, Svalbard (78°54′N, 12°13′E), seven kilometers southeast of Ny-Ålesund, Norway. Kittiwakes are colonial seabirds that breed on cliffs throughout the northern parts of the Pacific and

Atlantic, including the Barents Sea region up to the Svalbard Archipelago (Anker-Nilssen et al., 2000). Kittiwakes were studied in one plot of around 150 pairs breeding on cliff ledges at heights of 5–10 m. Male and female kittiwakes were sampled once, between 2007 to 2010 years, during the pre-laying stage (arrival, nest building, courtship and mating period) from 23rd of April to 16th of June. Table 1 summarizes sampling information: a total of 105 kittiwakes were sampled for measurement of Hg and 138 kittiwakes for POPs. We chose to focus our study on the pre-laying period, because sampling kittiwakes during the incubating or chick-rearing period would have biased our demographic study towards good-quality birds (breeders) and would have missed possible effects in non-breeders.

2.2. Capture and blood sampling

Male and female kittiwakes were caught on the nests with a noose at the end of a 5 m fishing rod. Blood samples were collected from the alar vein with a 2 ml heparinized syringe and a 23-gauge needle. Kittiwakes were individually marked with metal rings and PVC plastic bands engraved with a three-digit code and fixed to the bird's tarsus for identification from a distance without perturbation.

2.3. Laboratory analyses

Blood samples were centrifuged. Plasma and red blood cells were separated and stored at − 20°C. Molecular sexing was performed on red blood cells as detailed in Weimerskirch et al. (2005).Total Hg was measured at the laboratory Littoral Environnement et Sociétés (LIENSs) from lyophilized red blood cells with an Advanced Mercury Analyzer spectrophotometer (Altec AMA 254). At least two aliquots ranging from 5 to 10 mg dry weight were analyzed for each individual until having a relative standard deviation <5 %. As described by Bustamante et al. (2006), accuracy was checked using a certified reference material (CRM, Tort-2 Lobster Hepatopancreas, NRC, Canada; certified Hg concentration: 0.27 ± 0.06 µg g-1 dry mass; with recoveries of 98 to 102%). Mass of CRM was adjusted to represent the same amount of Hg introduced in the AMA compared to that in blood samples. Blanks were analysed at the beginning of each set of samples and the detection limit of the method was 0.005 µg g-1 dry mass. Mean values of replicates were used in statistical analyses.

POPs were analysed from whole blood samples at the Norwegian Institute for Air Research (NILU) in Tromsø. The following compounds were analysed: polychlorinated biphenyl (CB, -99, -118, -138, -153, -180, -183 and -187) hereafter referred as ∑ PCBs, p,p′-DDE (p,p'-dichlorodiphenyldichloroethylene, HCB (hexachlorobenzene), and the chlordane mixture (trans-chlordane, trans-, cis-nonachlor) and metabolites (oxychlordane), hereafter referred as CHL. To a blood sample of 0.5 to 1.5 ml, an internal standard solution was added (13C-labelled compounds from Cambridge Isotope Laboratories: Woburn, MA, USA). The sample was extracted twice with 6 ml of *n*-hexane, after denaturation with ethanol and a saturated solution of ammonium sulphate in water. Matrix removal on florisil columns, separation on an Agilent Technology 7890 GC and detection on an Agilent Technology 5975C MSD were performed as described by Herzke et al. (2009). The limit for detection was threefold the signal-to-noise ratio, and for the compounds investigated the limit ranged from 0.4 to 122 pg.g^{-1} wet weights (ww). For quality assurance, blanks (clean and empty glass tubes treated like a sample) were run for every 10 samples similar to standard reference material (1589 a human serum from NIST). The accuracy of the method was within the 70 and 108% range.

2.4. Life history traits

From 2007 to 2012, individuals were individually identified, through PVC plastic bands reading. Using a mirror at the end of an 8 m fishing rod, we checked the whole plot (about 120 nests) every two days to monitor breeding status (at least one egg is laid or no egg laid). Then, we checked the nest content every 2 or 3 days to monitor the number of chicks that reached at least 12 days of age per nest.

2.5. Statistical analyses

We used R software (R Development Core Team 2012) and generalized linear models (GLMs) with normal distribution and a link function to test whether log-transformed Hg, ∑ PCBs, DDE, HCB or CHL levels were linked to sex, year and the interaction sex × year. GLMs with binomial error distribution and a logit link function were then used to test whether breeding probability (will breed or will skip) the year of sampling was linked to pre-laying Hg, ∑ PCBs, DDE, HCB or CHL levels.

2.6. Estimating the effect of Hg and POPs on demographic parameters

The effects of Hg and POPs concentrations on the demographic parameters were evaluated through the capture-recapture data of sampled kittiwakes. A MSMR (Multi-State Mark Recapture, Lebreton and Pradel, 2002) model was constructed by distinguishing five states: non-breeder (NB, defined as an individual that was not observed with an egg), failed breeder (FB, defined as an individual that was observed with one or two eggs, or one or two chicks but that failed to raise a chick), successful breeder with one chick (SB1, defined as an individual that raised one chick), successful breeder with two chicks (SB2, defined as an individual that raised two chicks), and dead. The state dead (†) was an absorbing state representing death or permanent emigration from the study area. Kittiwakes that were ringed and observed the years before sampling for Hg or POPs were considered as non-observed, in order to test the effect of contaminants (at year t) on future (year t+1) survival and breeding performances. Models were parameterized in terms of the probability of survival (*S*), the probability of breeding (β), the probability of breeding successfully (γ), the probability of successfully raising two chicks (δ), and the detection probability (p). Transition probabilities between states were thus modeled with a four-step procedure where S, β, γ and δ were considered as four successive steps in transition matrices. Figure 1 presents a multinomial tree diagram describing the probability structure for multistate observations, and parameters of the model are defined in Table 2. We chose a MSMR approach since this allows taking into account the probability of detecting individuals given their return to the study sites. It also allows taking into account the previous breeding state of individuals which might be important to obtain unbiased estimates of demographic parameters (Lebreton and Pradel 2002).

Several constraints were made to ensure that the parameters of the model were estimable. The state "dead" being explicitly included in the model but being never encountered, transition probabilities from the state dead were fixed to 0 and capture probability was fixed to 0 (Pradel 2005, Choquet et al. 2009a). Because our capture-recapture analyses relied on a limited number of individual capture histories, parameters S, β, γ, δ and p were constrained to be constant over time but state and sex dependent. With this constraint the initial model was full-rank. Note that we ran a model where all demographic parameters were time, sex and state dependent but this model was highly rank deficient.

This MSMR model was parameterized by the survival–transition probabilities matrix:

$$\begin{array}{c} \\ \mathrm{NB} \\ \mathrm{FB} \\ \mathrm{SB1} \\ \mathrm{SB2} \\ \dagger \end{array} \begin{array}{c} \begin{array}{ccccc} \mathrm{NB} & \mathrm{FB} & \mathrm{SB1} & \mathrm{SB2} & \dagger \end{array} \\ \begin{bmatrix} S(1-\beta) & S\beta(1-\gamma) & S\beta\gamma(1-\delta) & S\beta\gamma\delta & * \\ S(1-\beta) & S\beta(1-\gamma) & S\beta\gamma(1-\delta) & S\beta\gamma\delta & * \\ S(1-\beta) & S\beta(1-\gamma) & S\beta\gamma(1-\delta) & S\beta\gamma\delta & * \\ S(1-\beta) & S\beta(1-\gamma) & S\beta\gamma(1-\delta) & S\beta\gamma\delta & * \\ - & - & - & - & * \end{bmatrix} \end{array}$$

Because we were interested to test for sex-specific effects of Hg and POPs on demographic parameters we started from an initial model including an effect of sex (*g*) on each parameter. Model selection was first performed on detection probability by testing state-dependency (difference between all states, between breeders and non-breeders, or no difference). We then tested for sex difference and state-dependency (difference between all states, difference between breeders and non-breeders or no difference) for *S, β, γ* and *δ*. We tested for an effect of Hg, ∑ PCBs, DDE, HCB, or CHL on demographic parameters the following year to test the hypothesis that contamination levels in one breeding season may influence the survival and breeding success of an individual in the following season. We built MSMR models where each demographic parameter θ was modeled as a function of contaminant *C* using a logit link function: $\mathrm{logit}(\theta) = a + b \times C_i$, where *a* is an intercept, *b* is a slope and C_i is Hg or POPs concentration for individual *i*. The 95% confidence interval (CI) of the slope parameters *b* was used, as well as Akaike's Information Criterion corrected for small sample size (AICc, Burnham and Anderson, 2002) for inference. We considered an effect of contaminant as statistically supported when 0 was outside the 95% CI of the mean of the slope of the relationship (Grosbois et al., 2008). When $b < 0$, or $b > 0$, the covariate *C* has a negative or positive effect on the demographic parameter, respectively. We tested the goodness-of-fit (GOF) of the time dependent MSMR model using U-CARE (Choquet et al. 2009b). All models were run under program E-SU RGE 1.8.5 allowing splitting transition probabilities between states (Choquet et al. 2009a).

3. Results

3.1 Associations between Hg or POPs and breeding probability in year of blood sampling

Table 1 summarizes the values of Hg, ∑ PCBs, DDE, HCB and CHL in males and females. Appendix 1 gives the concentrations of each POP congener and appendix 2 presents the relationships between levels of Hg, ∑ PCBs, DDE, HCB and CHL.

Hg levels were significantly higher in males than in females ($F_{1,103} = 3.993$, $p = 0.048$), but did not differ between the two sampling years (year: $F_{1,102} = 3.339$, $p = 0.071$; sex × year: $F_{1,101} = 1.102$, $p = 0.296$). Breeding probability during the sampling year was influenced by Hg levels ($df = 103$, $\chi^2 = 12.983$, $p < 0.001$): kittiwakes that would skip (mean ± SD: 2.284 ± 0.417 $\mu g.g^{-1}$) had higher pre-laying Hg levels than kittiwakes that would breed (1.962 ± 0.470 $\mu g.g^{-1}$).

Levels of ∑ PCBs, DDE, HCB, or CHL did not differ between males and females (sex: $p > 0.07$ for all tests: sex × year: $p > 0.09$ for all tests). Levels of ∑ PCBs ($F_{3,134} = 4.935$, $p = 0.003$), HCB ($F_{3,134} = 37.035$, $p < 0.001$), ∑ CHL ($F_{3,134} = 12.818$, $p < 0.001$), but not DDE ($F_{3,134} = 2.519$, $p = 0.061$) differed among years. Breeding probability was not influenced by levels of ∑ PCBs, DDE, HCB, or CHL during the sampling year ($p > 0.61$ for all tests).

3.2. Associations between Hg and demographic parameters in year after blood sampling

The GOF of the MSMR model was overall not significant (males: $\chi^2 = 48.913$, $df = 69$, $p = 0.968$ and females: $\chi^2 = 47.435$, $df = 71$, $p = 0.986$). The best model according to AICc (model 16, Appendix 3) indicated that breeders in the previous year had higher breeding probabilities and detection probabilities than non-breeders in the previous year. However birds captured as breeders or non-breeders did not differ in survival rate, probabilities of successfully raising one or two chicks (Appendix 3 and Table 3). Demographic parameters did not differ between males and females (Appendix 3 and Table 3).

Model selection and slope estimates suggested no effect of Hg on demographic parameters. Model Hg3 had a ΔAICc lower than 2 compared to the null model, but the effect of Hg on breeding probability the following year was not supported, since the 95% CI of the slope parameter included 0 (Table 4).

3.3. Associations between POPs and demographic parameters in year after blood sampling

Model selection was based on ΔAICc higher than 2 compared to the intercept model and the 95% CI of the slope of the relationship that did not include zero. Hence, in spite of good AICc, several models suggesting an effect of Σ PCBs, DDE, HCB or CHL on demographic parameters were not retained. Only six models met these requirements (Table 5). Models HCB5 and HCB6 suggested a negative effect of HCB on breeding probability the following year for individuals and especially males observed as breeders (Fig.2A, 2B). Model CHL6 suggested a negative effect of CHL on breeding probability the following year for males observed as breeders (Fig. 2C). Model HCB1 suggested a positive effect of HCB on the probability of successfully raising two chicks the following year (Fig. 3). Model HCB8 suggested a negative effect of HCB on survival rate of females (Fig. 4A). Model CHL7 suggested a negative effect of CHL on survival rate (Fig. 4B). We could also notice a tendency towards a negative effect between survival rates and levels of ∑ PCBs (model PCB7, ΔAICc = 1.24, mean slope and 95% CI = -0.44 [-0.82 ; -0.03]), DDE (Model: DDE7, ΔAICc = 0.88, slope = -0.42 [-0.82 ; -0.01]), HCB for males and females (Model HCB7, ΔAICc = 1.73, slope = -0.47 [-0.88 ; -0.06]), or CHL for females only (Model CHL8, ΔAICc = 1.50, slope = -0.73 [-1.29 ; -0.17]).

4. Discussion

Using a long-term data set and MSMR models, this study explores the demographic effects of Hg or families of legacy POPs (7 PCB congeners, p-p' DDE, HCB, and the chlordane mixture and metabolites (trans-chlordane, trans-, cis-nonachlor, oxychlordane)) in a free-living Arctic seabird species. It should be noticed that differences in toxicity among POP congeners were not taken into account in these analyses, because toxic equivalent factors (TEFs) were only available for PCB-105 and PCB-118. Moreover interactions among families of pollutants may occur within an organism to induce synergistic effects, but they are difficult to demonstrate within a field study.

4.1. Survival and contaminants

Estimated demographic parameters were similar to those previously estimated in other populations of black legged kittiwakes (Frederiksen et al., 2005). Adult survival rate in this study (85% [82 – 88%]) was within the range of estimated survival rates in north Atlantic populations (80-92%, Danchin and Monnat, 1992; Erikstad et al., 1995; Oro and Furness, 2002; Frederiksen et al., 2005).

The adult survival rate of kittiwakes was not jeopardized by Hg, which corroborates most of the previous studies in free-living birds (Wayland et al., 2008; Hallinger et al., 2011; Goutte et al., 2014a,b). Apparent survival rate was negatively linked to HCB levels in females, to mixture of chlordane and oxychlordane, and tended to be negatively correlated with ∑ PCBs or DDE levels. Only one study (Erikstad et al. 2013) highlighted a negative effect of oxychlordane on adult survival rate in the glaucous gull breeding in the Bjørnøya Island (blood levels of oxychlordane: 1.3 to 128.8 ng.g^{-1} wet weight, median: 13.2 ng.g^{-1} ww) and this effect was the most pronounced among the most contaminated females. Even if kittiwakes were more than 10-time less contaminated than glaucous gull (blood levels of oxychlordane: 0.007 to 6.0 ng.g^{-1} wet weight), this study reveals that high levels of the chlordane mixture and metabolites or HCB could negatively affect adult survival rate, and especially in female kittiwakes.

The correlation between POP levels and survival rate could be a by-product of age-dependent mechanisms, with older kittiwakes having the highest POP burden and the lowest survival probability. Age of kittiwakes was unknown in this study and we could not control for age. However, blood levels of PCB-153, p,p'-DDE, HCB, and oxychlordane were unrelated to age in glaucous gulls (Bustnes et al., 2003b). Similarly, blood levels of PCBs or organochlorine pesticides (HCB, lindane, chlordane mixture, mirex, DDT and metabolites) were unrelated to age in wandering albatrosses (Carravieri et al., 2014). Therefore, it seems unlikely that age was a confounding factor in the correlation between POP levels and survival rate. In addition, as we did not monitor long-distance dispersal, our findings on apparent survival rate could also include the effects of POPs on long-term emigration of the most polluted birds.

This study suggests that HCB or the chlordane mixture and metabolites may weaken the general health of kittiwakes and may increase their vulnerability to harsh environmental pressures in the Arctic (Letcher et al., 2010). In that context, it is conceivable that the effect of POPs on survival rate is only detected during harsh environmental events. Because our sample size did not allow taking into account an effect of years, we could not have tested whether harsh environmental conditions during a specific year would exacerbate the effects of pollutants on demographic parameters the following year.

4.2. Long-term fecundity and contaminants

A previous study on this population of kittiwakes has highlighted that total blood Hg load during the pre-laying period predicted the likelihood of breeding, with non-breeders having higher Hg levels than breeders, but not the timing of breeding, clutch size, and breeding success (Tartu et al., 2013). Moreover experimentally elevated Hg levels (total Hg in blood, mean ± SD: from 0.73 ± 0.09 to 3.95 ± 0.68 $mg.kg^{-1}$ fresh weight) led to an altered pairing behaviour in white ibises *Eudocimus albus* (Frederick and Jayasena, 2011). In the present study, Hg levels were higher in kittiwakes that would skip breeding than in birds that would breed, as previously shown (Tartu et al., 2013). Hg levels did not affect breeding probability and breeding success the following year, which differed from previous studies in the south polar skua *Catharacta maccormicki* (Hg levels in blood: mean ± SE: 2.15 ± 0.17 $\mu g.g^{-1}$ dry mass), in the brown skua *C. lonnbergi* (8.22 ± 0.24 $\mu g.g^{-1}$ dry mass) and in the wandering albatross (7.7 ± 3.6 $\mu g.g^{-1}$ dry mass) (Goutte et al., 2014 a,b). However, Hg levels in these species were measured during the incubation and the chick-rearing period, while Hg levels in the present study were measured in pre-laying kittiwakes. Furthermore, breeding success was monitored on chicks that reached at least 12 days of age and did not allow testing an effect of contaminants on late developmental stage.

POPs burden did not influence the breeding probability the year of sampling, which was consistent with a previous study on the same population of kittiwakes (Tartu et al. 2014). Breeding probability the following years was reduced by high HCB levels in breeders and especially in males, or by high levels of the chlordane mixture and metabolites in male breeders. A negative correlation between POP levels and breeding probabilities the following year has been highlighted in the wandering albatross (Goutte et al., 2014b). Male breeders seemed to be the most sensitive to POPs. Energetic and time-dependent costs of reproduction have been shown to induce downstream consequences on reproductive investment during the following breeding season (carry over effect, Catry et al., 2013). One may suggest that POPs burden may intensify these carry over effects, but studies are needed to either rebut or confirm this hypothesis.

Levels of ∑ PCB, DDE, HCB, and the chlordane mixture and metabolites did not influence the probability of successfully raising one chick the following year, which was consistent with a previous study on the same population of kittiwakes and during the year of sampling (Tartu et al., 2014). We detected a positive relationship between the probability of successfully raising two chicks the following year and HCB levels, but not PCBs, DDE or the chlordane mixture. This positive relationship between HCB and breeding performance appears surprising, as contaminants are believed to induce deleterious effects on reproductive traits. Previous studies have pointed out that female kittiwakes and gulls with higher levels of organochlorine pesticides laid their eggs earlier in the season (Bustnes et al., 2008; Tartu et al., 2014). As laying early is related to high breeding success (Lack, 1968), this could explain the positive relationship between HCB and the probability of successfully raising two chicks. In another hand, this relationship may not be causal and may be enhanced by confounding factors: for instance, kittiwakes succeeding in raising two chicks may be of higher quality, rely on higher trophic level organisms and hence be more exposed to pollutant.

It appears that some families of POPs may be more prone to trigger damaging effects the following year. Specifically, high levels of HCB or the chlordane mixture and metabolites were correlated to lower survival rate and lower probability to breed the following year. These findings corroborate a previous study: despite their lower concentrations, HCB and oxychlordane tended to be more often related to adverse effects than PCB and DDE in glaucous gull (Bustnes, 2006). Although levels of these "legacy" POPs are expected to decline, as shown in Canadian Arctic seabirds from the 1970s to the late 1990s (Braune et al., 2005), they appear to represent a potential threat for adult survival rate and thus for population dynamics.

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Table 1: Levels (mean ± SD) of Σ PCBs (CB, -99, -118, -138, -153, -180, -183 and -187), p,p′-DDE, HCB, CHL (transchlordane, trans-, cis-nonachlor, oxychlordane,) and Hg (mercury) in blood of male and female kittiwakes sampled during the pre-laying period.

	Year	Males	Females
Σ PCBs	2007	14700 ± 9630	12640 ± 6421
(pg.g^{-1} ww)	2008	14896 ± 11029	13399 ± 9197
	2009	9282 ± 7915	10375 ± 4705
	2010	12786 ± 10966	21168 ± 14390
DDE	2007	3622 ± 1730	3152 ± 1422
(pg.g^{-1} ww)	2008	4025 ± 2642	4189 ± 3490
	2009	2618 ± 1660	2184 ± 890
	2010	3249 ± 2739	4725 ± 3584
HCB	2007	1616 ± 966	1600 ± 407
(pg.g^{-1} ww)	2008	1616 ± 444	1691 ± 697
	2009	2416 ± 1493	2699 ± 451
	2010	2670 ± 877	3487 ± 1288
CHL	2007	1352 ± 782	1329 ± 508
(pg.g^{-1} ww)	2008	1237 ± 510	1275 ± 765
	2009	1344 ± 1155	1353 ± 403
	2010	1766 ± 650	2482 ± 1602
Hg	2008	2.06 ± 0.44	1.97 ± 0.44
(µg.g^{-1} dw)	2009	2.33 ± 0.55	2.01 ± 0.41

Table 2 Definition of parameters used in the multistate mark–recapture model

Parameter	Definition
$S^t{}_s$	Probability that an individual in state *s* at time t survives to time *t + 1* and does not permanently emigrate from the study area
$\beta^t{}_s$	Probability that an individual in state *s* at time t breeds at time *t + 1* given that it survives to *t + 1*
$\gamma^t{}_s$	Probability that an individual in state s at time t breeds successfully at time t + 1 given that it survives to and breeds at time t + 1
$\delta^t{}_s$	Probability that an individual in state *s* at time *t* raises successfully two chicks at time *t + 1* given that it survives to and breeds successfully at time *t + 1*
$p^t{}_s$	Probability that an individual in state *s* at time t is encountered at time *t + 1*

Table 3: Estimation of parameters (mean and CI) calculated from the best model (model 16, Appendix 3) for breeders and non-breeders.

	Non-breeders	Breeders
S: apparent survival rate (%)	85 [82 ; 88]	85 [82 ; 88]
β: Breeding probability (%)	47 [41 ; 53]	82 [78 ; 86]
γ: Breeding success (%)	75 [71 ; 79]	75 [71 ; 79]
δ: Probability of raising 2 chicks (%)	40 [35; 45]	40 [35; 45]
p: Detection probability (%)	78 [67 ; 85]	98 [90 ; 99]

Table 4: Modeling the effects of Hg levels and sex on demographic parameters of *Rissa tridactyla* (N = 105). Models are arranged from lowest to highest ΔAICc. The estimated slope and 95% confidence intervals (CI) are given for the model (Hg3) that has a lower AICc than the intercept model.

Hypothesis	# Model	Rank	Deviance	ΔAICc	Slope	95% CI
Effect of Hg on γ	Hg3	12	1194.84	0	0.29	-0.84 ; 1.43 #
Intercept model	Hg0	10	1201.22	2.10		
Effect of Hg on δ	Hg1	12	1197.40	2.56		
Effect of Hg and sex on γ	Hg4	14	1193.67	3.18		
Effect of Hg and sex on δ	Hg2	14	1194.82	4.33		
Effect of Hg on S	Hg7	12	1201.11	6.28		
Effect of Hg on β	Hg5	14	1197.90	7.41		
Effect of Hg and sex on β	Hg6	18	1190.66	9.03		
Effect of Hg and sex on S	Hg8	14	1200.50	10.01		

This effect is not supported because the 95% confidence intervals of the mean of the slope of the relationship included zero.

Table 5: Modeling the effects of Σ PCBs, p,p′-DDE, HCB and CHL levels and sex on demographic parameters of *Rissa tridactyla* (N = 138). Models are arranged from lowest to highest ΔAICc. The estimated slopes and 95% confidence intervals (CI) are given for models that have a lower AICc than the intercept model (NB: non-breeders, B: breeders).

Hypothesis	# Model	Rank	Deviance	ΔAICc	Slope	95% CI	
Effect of Σ PCBs on β	PCB5	14	1351.03	0	NB : -0.62 B : -0.14	-1.48 ; 0.23 -0.82 ; 0.53	# #
Effect of Σ PCBs and sex on β	PCB6	18	1344.22	1.90	Male NB : -0.36 Male B : -1.10 Female NB : -0.87 Female B : 0,83	-1.50 ; 0.78 -2.43 ; 0.22 -2.21 ; 0.46 -0.38 ; 2.06	# # # #
Effect of Σ PCBs on S	PCB7	12	1366.07	10.75	-0.44	-0.82 ; -0.03	##
Effect of Σ PCBs on δ	PCB1	12	1367.05	11.74	0.47	-0,36 ; 1,31	###
Intercept model	PCB0	10	1371.55	11.99			
Effect of Σ PCBs and sex on δ	PCB2	14	1364.06	13.03			
Effect of Σ PCBs and sex on S	PCB8	14	1364.33	13.30			
Effect of Σ PCBs on γ	PCB3	12	1368.92	13.61			
Effect of Σ PCBs and sex on γ	PCB4	14	1368.69	17.66			
Effect of DDE and sex on β	DDE6	18	1339.67	0	Male NB : -0.26 Male B : -1.17 Female NB : -1.82 Female B : 0.69	-1.78 ; 1.26 -2.44 ; 0.10 -3.79 ; 0.14 -0.64 ; 2.01	# # # #
Effect of DDE on β	DDE5	14	1349.00	0.61	NB : -1.00 B : -0.14	-2.08 ; 0.08 -0.70 ; 0.42	# #
Effect of DDE on S	DDE7	12	1366.43	13.76	-0.42	-0.82 ; -0.01	##
Intercept model	DDE0	10	1371.55	14.64			
Effect of DDE on δ	DDE1	12	1367.91	15.24			
Effect of DDE on γ	DDE3	12	1368.85	16.18			
Effect of DDE and sex on S	DDE8	14	1365.94	17.56			
Effect of DDE and sex on δ	DDE2	14	1366.73	18.35			
Effect of DDE and sex on γ	DDE4	14	1368.19	19.80			
Effect of HCB and sex on β	HCB6	18	1339.36	0	Male NB : -1.50 Male B : -1.86 Female NB : -0.02 Female B : 0.08	-4.24 ; 1.25 -3.38 ; -0.34 -0.87 ; 0.92 -0.74 ; 0.90	# # #

Effect of HCB on β	HCB5	14	1349.44	1.37	NB : -0.28 B : -0.53	-1.06 ; 0.50 -1.04 ; -0.01	#
Effect of HCB and sex on δ	HCB2	14	1357.05	8.98	NB : -0.18 B : -2.27	-0.57 ; 0.21 -0.15 ; 4.69	# #
Effect of HCB on δ	HCB1	12	1362.22	9.86	0.94	0.10 ; 1.79	
Effect of HCB and sex on S	HCB8	14	1360.54	12.47	Male : 0.41 Female : -0.82	-0.75 ; 1.57 -1.39 ; -0.25	#
Effect of HCB on S	HCB7	12	1365.58	13.22	-0.47	-0.88 ; -0.06	##
Intercept model	HCB0	10	1371.55	14.95			
Effect of HCB on γ	HCB3	12	1369.19	16.83			
Effect of HCB and sex on γ	HCB4	14	1367.08	19.01			
Effect of CHL and sex on β	CHL6	18	1338.80	0	Male NB : -0.59 Male B : -2.64 Female NB : -0.73 Female B : -0.07	-2.95 ; 1.77 -5.09 ; -0.18 -1.98 ; 0.51 -0.85 ; 0.70	# # #
Effect of CHL on β	CHL5	14	1347.61	0.10	NB : -0.73 B : -0.59	-1.81 ; 0.34 -1.20 ; 0.01	# #
Effect of CHL on S	CHL7	12	1363.00	11.20	-0.57	-1.00 ; -0.13	
Effect of CHL and sex on δ	CHL2	14	1359.59	12.08	NB : 1.46 B : 1.85	-1.23 ; 4.15 -0.23 ; 3.93	# #
Effect of CHL on δ	CHL1	12	1363.83	12.03	1.05	-0.15 ; 2.24	#
Effect of CHL and sex on S	CHL8	14	1361.52	14.01	Male : -0.04 Female : -0.73	-1.12 ; 1.04 -1.29 ; -0.17	### ##
Intercept model	CHL0	10	1371.55	15.51			
Effect of CHL on γ	CHL3	12	1368.41	16.61			
Effect of CHL and sex on γ	CHL4	14	1368.38	20.87			

This effect is not supported, because the 95% CI of the mean of the slope of the relationship included zero.

This effect is not supported, because the model has a ΔAICc < 2 compared to the intercept model

This effect is not supported, because the 95% CI of the mean of the slope of the relationship included zero and the model has a ΔAICc < 2 compared to the intercept model.

Figure 1: A multinomial tree diagram describing the probability structure for multistate observations. Solid boxes indicate the states alive in state NB (non-breeder), FB (failed breeder), SB1 (successful breeder with one chick), SB2 (successful breeder with two chicks). dead. State transition probabilities were decomposed in a four-step process. The state transitions *(S, β, γ, δ)* are defined in Table 2 and states in the Methods section.

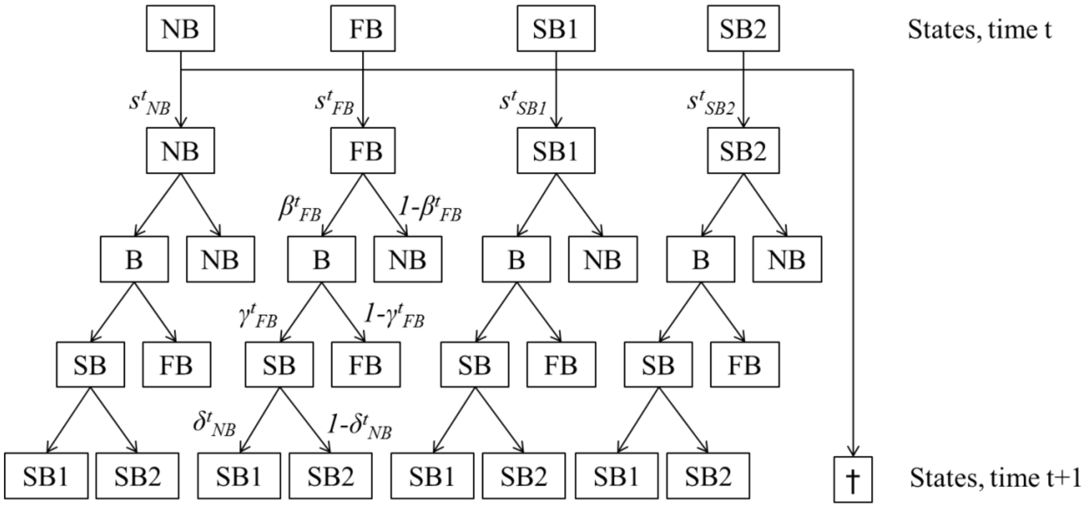


Figure 2: Relationship between breeding probability at year t+1 in black-legged kittiwakes and (A) standardized HCB levels in individuals observed as breeders at year t, (B) standardized HCB levels in males observed as breeders at year t and (C) standardized levels of the chlordane mixture and metabolites in males observed as breeders at year t . Dotted lines represent 95% CI.

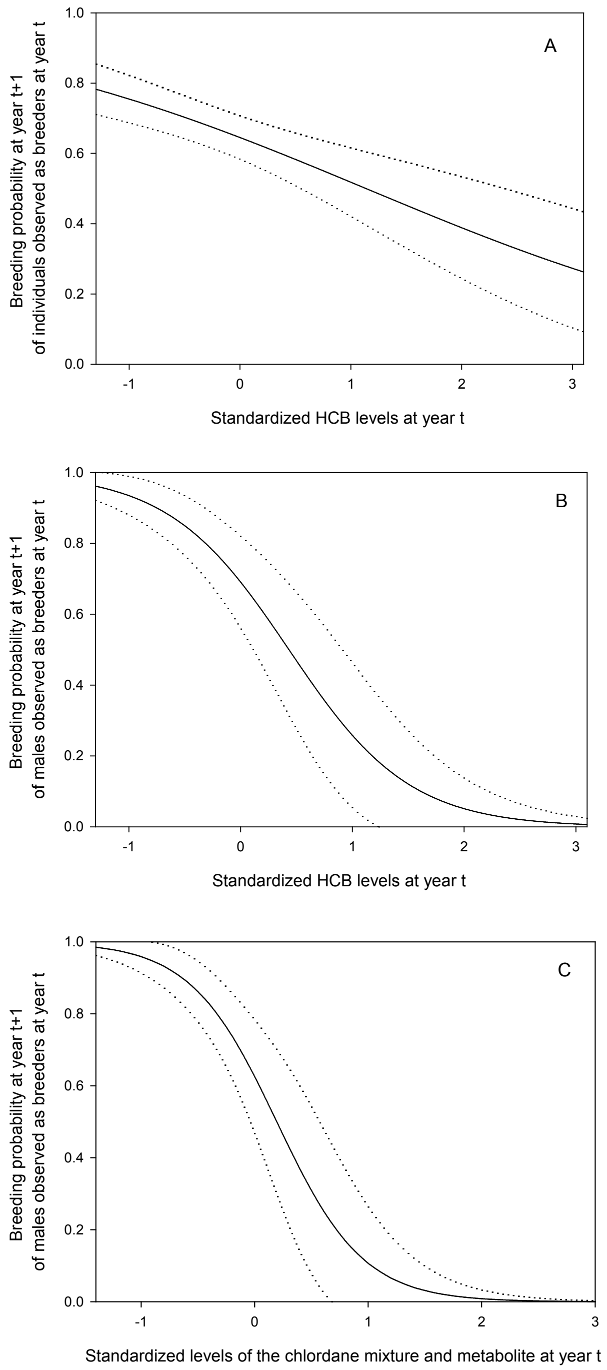


Figure 3: Relationship between probability of successfully raising two chicks at year t+1 in black-legged kittiwakes and standardized HCB levels at year t.

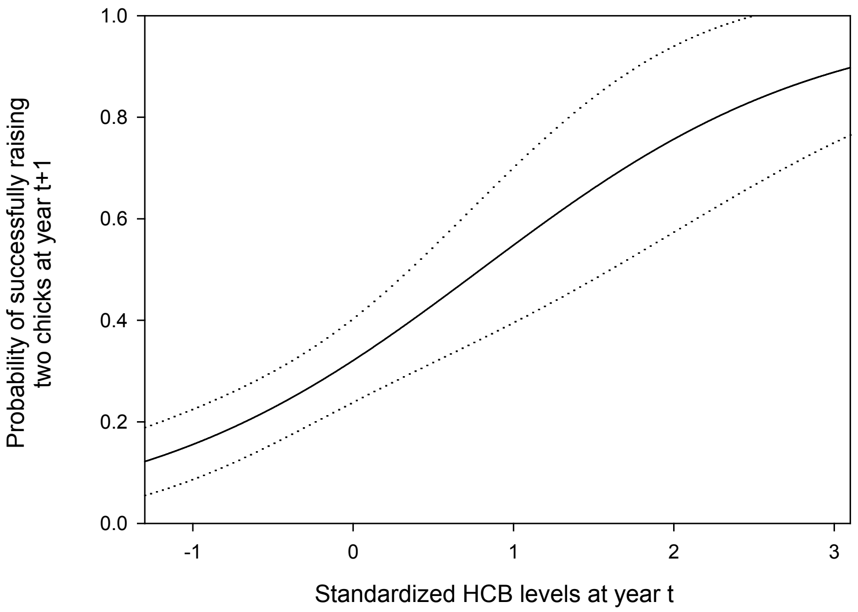
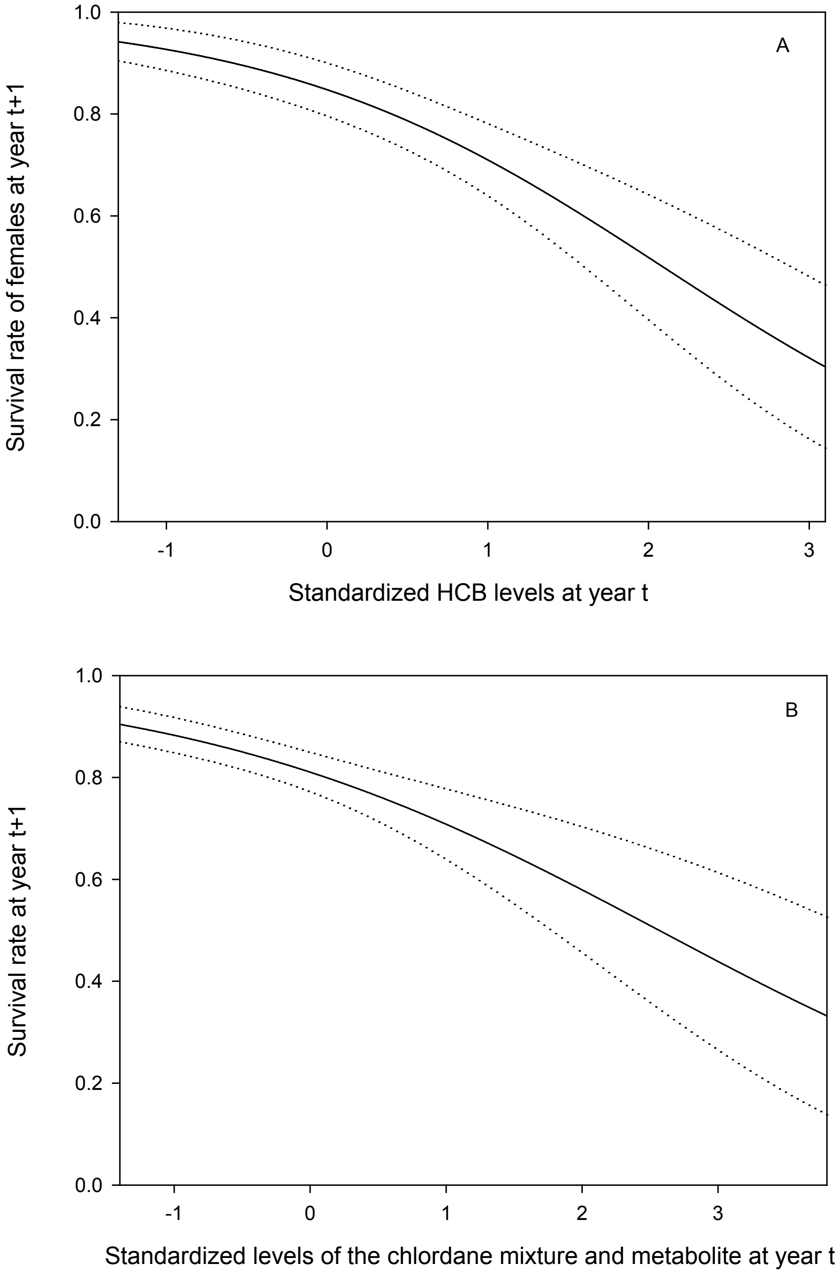


Figure 4: Relationship between survival rate at year t+1 in black-legged kittiwakes and (A) standardized HCB levels in females at year t, (B) standardized levels of the chlordane mixture and metabolites at year t.



Appendix 1: Concentrations (mean, median and standard deviation SD, in pg.g-1 ww) measured for each POP congener in 138 black-legged kittiwakes during the pre-laying period.

Congener	Mean	Median	SD
PCB-99	1069	800	869
PCB-118	1638	1313	1217
PCB-138	4104	2856	3463
PCB-153	5498	4159	3979
PCB-180	1888	1513	1832
PCB-183	537	354	591
PCB-187	932	680	867
p,p' DDE	3892	2978	2900
HCB	2418	2026	1262
transchlordane	322	217	346
oxychlordane	1002	818	786
cisnonachlor	173	158	100
transnonachlor	216	196	142

Appendix 2: Relationships between Hg, ∑ PCBs, DDE, HCB, and CHL levels

	∑ PCB	DDE	HCB	CHL
DDE	$F_{1,136} = 180$, $p < 0.001$			
HCB	$F_{1,136} = 63$, $p < 0.001$	$F_{1,136} = 41$, $p < 0.001$		
CHL	$F_{1,136} = 129$, $p < 0.001$	$F_{1,136} = 104$, $p < 0.001$	$F_{1,136} = 201$, $p < 0.001$	
Hg	$F_{1,35} < 0.01$ $p = 0.983$	$F_{1,35} = 0.04$ $p = 0.839$	$F_{1,35} = 2.77$ $p = 0.105$	$F_{1,35} = 1.33$ $p = 0.136$

Appendix 3:

Initial model (Model1) considers sex-difference and state-difference on *S, β, γ, δ*, and *p*. A. Modelling the effect of states on *p*. B. Modelling the effect of sex on *s, β, γ* and *δ*, with p being different between breeders and non-breeders. C. Modelling the effect of states on *s, β, γ* and *δ* (*δ* is necessarily constant).

A. Hypothesis	# Model	Rank	Deviance	ΔAICc
***p* differs between NB and B**	**Model2**	**31**	**3630,77**	**0**
p is state-dependent	Model1	33	3630,23	3,73
p is constant	Model3	30	3642,96	10,06

B. Hypothesis	# Model	Rank	Deviance	ΔAICc
No sex difference on *S, β, γ* and *δ*	**Model8**	**18**	**3636,86**	**0**
Sex difference on *S, β* and *δ*	Model5	27	3632,09	14,08
Sex difference on *S, γ* and *δ*	Model6	27	3632,90	14,88
Sex difference on *β, γ* and *δ*	Model7	27	3633,21	15,19
Sex difference on *S, β* and *γ*	Model4	30	3630,77	19,11
Sex difference on *S, β, γ* and *δ*	Model2	31	3630,77	21,24

C. Hypothesis	# Model	Rank	Deviance	ΔAICc
S and *γ* are constant; *β* is state-dependent	Model15	12	3640,14	0
***S* and *γ* are constant; *β* differs between NB and B**	**Model16**	**10**	**3644,82**	**0,59**
S, β and *γ* are constant	Model17	11	3644,79	2,61
γ is constant; *S* and *β* are state-dependent	Model10	15	3638,35	4,38
S is constant; β and *γ* are state-dependent	Model14	15	3638,68	4,72
S differs between NB and B; β and *γ* are state-dependent	Model13	16	3637,55	5,64
S, β and *γ* are state-dependent	Model8	18	3636,86	9,10
β differs between NB and B; *S* and *γ* are state-dependent	Model11	16	3641,59	9,69
γ differs between NB and B; *S* and *β* are state-dependent	Model9	16	3678,02	46,12
β is constant; *S* and *γ* are state-dependent	Model12	15	3725,68	91,71