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Temporal trends of POPs in arctic foxes from Svalbard in light of a changing climate

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Leiv Eiriksson
Svart skip på grått hav.
Dagar og nætir under segl.
Uviss er makta til Ægir.
Kanskje tek havet aldri slutt?
Likevel stemner du vest,
lit på at hav klappar mot land.

Olav H. Hauge

Takksigelser

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Martin Andersen

Abstract

The present study investigates concentrations and temporal trends of persistent organic pollutants (POPs) in arctic foxes (*Vulpes lagopus*) from Svalbard, Norway, adjusted for sex, age, body condition, diet, reindeer mortality and sea ice coverage. Number of reindeer carcasses in Adventdalen and sea ice coverage of Isfjorden in the spring preceding the trapping season were used as indexes for climate influenced food availability between years. We analysed liver of 100 foxes from Svalbard, collected in 1997/98, 1998/99, 1999/00, 2001/02, 2002/03, 2003/04 and 2010/11 for concentrations of polychlorinated biphenyls (PCBs; PCB -28, -52, -101, -118, -138, -153 and -180), chlordanes (*cis*-chlordane, trans-nonachlor and oxychlordane), *p-p'*-DDT, *p-p'*-DDE, HCB, mirex and β-HCH. The POPs found in highest concentrations were oxychlordane, PCB-180 and PCB-153. We found evidence for a temporal decrease in ∑PCBs (PCB -118, -138, -153, -180), and ∑chlordanes (trans-nonachlor and oxychlordane) when controlling for possible confounding variables. We also found evidence for an effect of body condition and $\delta^{13}C$ on the POP concentrations, as thinner foxes and foxes feeding from the marine food web had significantly higher levels of POPs. There was no evidence for effects of sex, age, reindeer mortality and sea ice coverage on the concentrations of POPs, although increased reindeer mortality had a non-significant negative effect on all the POPs analysed. This study shows that correcting for body condition and diet is vital when investigating temporal trends of POPs in biota. It also illustrates some of the difficulties of investigating POP concentrations in an arctic predator living in an environment influenced by climatic variations.

Contents

1. Introduction 1

2. Material and methods 4

2.1 Sample collection 4

2.2 Reindeer mortality data 5

2.3 Ice data 5

2.4 Stable isotope analysis of $\delta^{13}C$ and $\delta^{15}N$ 5

2.5 Chemical analysis of OCPs and PCBs 6

2.6 Data analysis: 8

3. Results 10

3.1 POP concentrations 10

3.2 Factors underlying contaminant trends 11

4. Discussion 14

4.1 POP concentrations 14

4.2 Factors underlying contaminant trends 16

4.3 Relationship between $\delta^{13}C$, Reindeer mortality and sea ice cover 19

4.4 Influence of climate change on POP concentrations in arctic foxes from Svalbard 20

4.5 Conclusion and future perspectives 21

5. References 22

Appendix 1 28

Appendix 2 29

Appendix 3 30

Appendix 4 31

1. Introduction

Persistent organic pollutants (POPs) are a group of chemicals that were previously used extensively in a large variety of applications. Several of the compounds were designed to be toxic and were used as pesticides and fungicides, while others are products and by-products of industrial processes (AMAP, 1998; Jones and de Voogt, 1999). Because of their potential for long range transport, biomagnification, persistency in nature and toxic effects, many of the legacy POPs were banned under the Stockholm Convention in 2004 (Hop et al., 2002; Jones and de Voogt, 1999; Letcher et al., 2010; Sinkkonen and Paasivirta, 2000; Stockholm Convetion, 2004; Wania and Mackay, 1993). Despite this and several other legal restrictions in usage, POPs have continued to prevail in nature. Although temporal trends have revealed a general decrease in the level of contaminants in arctic biota, arctic apex predators still have tissue levels of contaminants which may lead to harmful effects (Dietz et al., 2013; Letcher et al., 2010; Rigét et al., 2010).

The arctic fox is one of the highest contaminated arctic apex predators (Fuglei et al., 2007; Klobes et al., 1998; Severinsen and Skaare, 1997). Individual contaminant loads in this species are hypothesised to have severe effects (Brunstrom and Halldin, 2000; Letcher et al., 2010; Sonne, 2010) and according to the recommendations of recent reports, arctic foxes should be included in future monitoring of contaminants in arctic biota (Gabrielsen et al., 2012; Kallenborn et al., 2011).

The arctic fox is a small canine species (ca. 3-5 kg), with a circumpolar distribution (Audet et al., 2002). Arctic foxes are classified into two different ecotypes; “lemming foxes” and “coastal foxes”. The lemming ecotype is present on inland tundra areas, closely connected to microtine rodents as an apex predator. The coastal ecotype inhabits islands and other regions where populations of microtine rodents are absent or limited, foraging as an opportunistic predator and scavenger from both the marine and terrestrial ecosystems (Angerbjorn et al., 2004; Braestrup, 1941). Svalbard arctic foxes belong to the coastal ecotype (Dalén et al., 2005; Fuglei et al., 2003; Prestrud, 1992). Within this population, fox habitats can be divided into resource areas differing in whether the main food sources are marine or terrestrial (Eide et al., 2005; Eide et al., 2012; Frafjord, 1993). The Svalbard arctic fox is a true opportunist with a potential for a highly variable diet throughout the year.

Independent of ecological distinctions, it will efficiently utilize the few resources available, both as a predator and scavenger, in order to ensure survival and reproduction (Angerbjorn et al., 1994; Eide et al., 2005; Pond and Gilmour, 1997; Prestrud, 1992).

The Svalbard reindeer (*Rangifer tarandus platyrhynchus*) and Svalbard rock ptarmigan (*Lagopus muta hyperborea*) are the only terrestrial resident species available as food items in both summer and winter on Svalbard. As such, these constitute a major part of the arctic fox diet (Eide et al., 2005; Eide et al., 2012; Frafjord, 2002; Fuglei et al., 2003). Additional food items for foxes during spring and summer are seabirds and geese (Eide et al., 2005; Eide et al., 2012; Frafjord, 1993; Prestrud, 1992). As migratory birds leave Svalbard by October, food sources might be scarce during winter, and previously cached food is thought to be important (Frafjord, 1993; Prestrud, 1992). Arctic foxes may also follow polar bears (*Ursus maritimus*) on the sea ice, scavenging remains of killed seals (*Phoca* sp.), or hunt ringed seal pups (*Pusa hispida*) in early spring (Gjertz and Lydersen, 1986; Hiruki and Stirling, 1989; Roth, 2002).

The possible effects of climate variability on arctic fox diet has been addressed several times, particularly through changes in the availability of reindeer carcasses and sea ice cover (Eide et al., 2012; Fuglei and Ims, 2008; Fuglei et al., 2003; Hansen et al., 2013; Pamperin et al., 2008). The Svalbard reindeer is strongly influenced by weather conditions, particularly in winter (Aanes et al., 2002; Aanes et al., 2003; Hansen et al., 2011; Hansen et al., 2013). Winters with frequent “rain-on-snow” events often results in a thick layer of ground ice and ice in the snow pack which makes forage resources inaccessible. This can lead to starvation causing reduced reproduction and high mortality rates in the Svalbard reindeer population (Hansen et al., 2011; Hansen et al., 2013; Tyler and Øritsland, 1999). Recent studies have found this seasonal access to vast amounts of reindeer carrions greatly influence the Svalbard arctic fox population (Eide et al., 2012; Hansen et al., 2013). Ringed seal pups are born inside a snow lair on the ice in April-May (Gjertz and Lydersen, 1986; Smith et al., 1991). Arctic foxes can hunt ringed seal pups on the sea ice, often with a higher success rate than polar bears (Smith and Lydersen, 1991). Over the last decade, there has been large variation in the sea ice cover inside several fjords on the western coast of Spitsbergen (Pavlov et al., 2013; Zhuravskiy et al., 2012). It is therefore expected that sea ice conditions in spring can influence the prey availability and access to food resources for arctic foxes (Pamperin et al., 2008; Roth, 2002).

There is a growing body of evidence indicating that variation in climate and feeding conditions strongly influence temporal trends of POPs in biota (McKinney et al., 2013; McKinney et al., 2009). Prey availability and feeding opportunities are critical considerations when investigating contaminant trends of wildlife predators (Bustnes et al., 2011; Gaden et al., 2012). Different methods can be used to investigate prey availability and dietary variation of arctic foxes. Traditionally, feeding habits have been studied using field observations, analyses of scats from abandoned dens and stomach contents (Eide et al., 2005; Frafjord, 1993; Prestrud, 1992). Lately, stable isotope analysis (SIA) has been confirmed as a useful tool in ecotoxicology, verifying that individuals feeding at a higher trophic position, usually from the marine food web, have increased levels of POPs (Fisk et al., 2001; McKinney et al., 2013). The ratio of stable isotopes of carbon ($^{13}C/^{12}C$; $\delta^{13}C$) is often used to distinguish between terrestrial and marine protein intake, while the ratio of nitrogen stable isotopes ($^{15}N/^{14}N$; $\delta^{15}N$) reflects the trophic position of an individual (Fisk et al., 2001; Hop et al., 2002; Kelly, 2000).

The first two studies on concentrations of ∑PCBs in Svalbard arctic foxes are from foxes captured in 1973/74 and 1983/84. These reported similar liver concentrations with means of 12400 and 9700 ng/g lipid weight (l.w.), respectively (Norheim, 1978; Wang-Andersen et al., 1993). Severinsen and Skaare (1997) found extremely high levels in liver tissues of foxes captured in 1991/92 (∑PCBs: 20466 ng/g l.w.), before Fuglei et al. (2007) reported a decrease in PCBs from adipose tissues of foxes caught in 1998/99 (∑PCBs: 9875 ng/g l.w.). High levels of ∑chlordanes have previously been found in livers of arctic foxes (Hoekstra et al., 2003a; Letcher et al., 2010; Severinsen and Skaare, 1997), with the chlordane metabolite oxychlordane dominating the quantitative amounts (Hoekstra et al., 2003a; Klobes et al., 1998). Mirex, *p,p′*-DDT and *p,p′*-DDE has, to our knowledge, previously been measured only one time in arctic foxes from Svalbard, at levels reaching 668, 156 and 639 ng/g l.w., respectively (Fuglei et al., 2007; Severinsen and Skaare, 1997).

The main objective of this study is to investigate temporal trends of POPs in arctic foxes from Svalbard, Norway, considering changes in food availability and feeding habits. We hypothesise that foxes feeding from the marine food web have higher levels of contaminants. We also hypothesise that climatic variation between years will affect the diet and levels of POPs in arctic foxes.

2. Material and Methods

2.1 Sample collection

Arctic foxes were trapped on Spitsbergen, Svalbard, mainly around the Isfjorden area, Nordenskiöld Land (77-79° N, 13-18° E; Figure 1). The foxes were collected by trappers, using baited traps during the annual harvest between 1st November and 15th March in 1997/98 (n=14), 1998/99 (n=14), 1999/00 (n=14), 2001/02 (n=14), 2002/03 (n=14), 2003/04 (n=14) and 2010/11 (n=16). The foxes were weighed, sex determined and skinned at Svalbard before the frozen carcasses were transported to Tromsø where the final dissection took place. The age of the foxes was determined by counting the annuli in the cementum of a sectioned canine tooth (Grue and Jensen, 1976). To evaluate body condition, a subjective fat index based on visual inspection of the skinned carcasses was used (Prestrud and Nilssen, 1992). The index ranged from 0 to 4 (none to extensive). Samples of skeletal muscle and liver were packed in aluminium foil and stored at -20°C until further analysis. All foxes in this study were between 1 and 2 years old. Svalbard arctic fox vixens do not reach maximum pregnancy rates before the age of four (Eide et al., 2012). We therefore expect none of the foxes to have given birth. A total of 100 foxes were analyzed and sex, age and body condition were balanced over years.

Figure 1: Map over the study area on Nordenskiöld Land on Spitsbergen, Svalbard

2.2 Reindeer mortality data

We used annual reindeer mortality (i.e. number of reindeer carcasses) as an index for availability of reindeer as food for arctic foxes in the spring preceding the trapping season. The reindeer mortality data was derived from a long-term monitoring survey in Adventdalen (Hansen et al., 2013; Tyler and Øritsland, 1999). The surveys were conducted in June/July by 6 observers recording the total number of reindeer carcasses in Adventdalen and side valleys (Hansen et al., 2013; Tyler and Øritsland, 1999). This way of monitoring reindeers on Svalbard is considered to be a reliable method of estimating reindeer winter mortality (Solberg et al., 2001; Tyler et al., 2008; Tyler and Øritsland, 1999). Due to the high correlations in population density among reindeers in different areas of Nordenskiöld Land (Aanes et al., 2003), we assumed that the inter annual pattern of reindeer mortality in Adventdalen was a good proxy for other populations in Nordenskiöld Land.

2.3 Ice data

We used sea ice coverage of Isfjorden as an index for availability of ringed seal pups as prey for arctic foxes during the spring preceding the trapping season. The total area (m^2) of Isfjorden covered by fast ice for more than 14 days in the period 15th March - 30th April were calculated from satellite images. The data does not represent more than 14 consecutive days of sea ice, but rather more than 14 total days with fast ice in the period.

2.4 Stable isotope analysis of $\delta^{13}C$ and $\delta^{15}N$

Stable isotope ratios of muscle provide dietary information from the previous 1-2 months (Roth and Hobson, 2000). Since muscle tissues were sampled from foxes trapped between November and March, SIA is expected to reflect the autumn/winter diet, depending on when the fox was caught.

Approximately 1g (cm^3) of muscle tissue from each individual was dried at 60°C for 2-3 days before it was grounded into a fine powder in a bead-mill homogenizer (TissueLyzerII, Qiagen Gmbh, Hilden, Germany) and 0.4mg (±0.05mg) was put into a tin container. Stable isotope analysis was performed at the Stable Isotopes in Nature Laboratory (SINLAB), New Brunswick, Canada. The combustion analyses were as described by Ehrich et al. (2011). In short, the samples were combusted in a Carlo Erba NC2500 Elemental Analyzer before delivery to a Finnigan MatDelta Plus mass spectrometer (Thermo Finnigan, Bremen, Germany). Stable isotope signatures are expressed as parts per thousand (‰) relative to a standard as follows: $[(R_{\text{sample}}/R_{\text{standard}})-1]\times 1000$, where R is the fractions of heavy to light isotopes ($\delta^{13}C/\delta^{12}C$ and $\delta^{14}N/\delta^{15}N$). Peedee belemnite carbonate and atmospheric nitrogen were used as standards for $\delta^{13}C$ and $\delta^{15}N$ respectively (Kelly, 2000). Due to different biochemical pathways in their synthesis, the stable isotope ratios of carbon is lower in lipids than in other tissues (DeNiro and Epstein, 1977); thus the $\delta^{13}C$ of a sample also reflects the lipid content of the tissue sample (Ehrich et al., 2011). The model based normalisation for muscular tissue proposed by Ehrich et al. (2011) was used to correct the value of $\delta^{13}C$ for lipid content in the samples with a C/N ratio between 3.5 and 7. Three samples were not corrected as these had a C/N ratio < 3.5 and were considered lean, none of the samples had a C:N ratio > 7.

2.5 Chemical analysis of OCPs and PCBs

The liver samples were analyzed for POPs at the Laboratory of Environmental Toxicology at the Norwegian School of Veterinary Science (NVH) in Oslo. The laboratory is accredited by Norwegian Accreditation for testing the analyzed chemicals in biological material according to the requirements of the NS-EN ISO/IEC 17025 (TEST 137). The following organochlorine pesticides (OCPs) and polychlorinated biphenyls (PCBs) were analysed; PCBs: PCB-28, -52, -101, -118, -138, -153 and -180 (ΣPCB7), *cis*-octachloro-4,7-methanohydroindane (*cis*-chlordane), *trans*-nonaklor, oxychlordane, dichlorodiphenyltrichloroethane (*p,p′*-DDT), dichlorodiphenyldichloroethylene (*p,p′*-DDE), hexachlorobenzene (HCB), perchloropentacyclodecane (mirex) and beta-hexachlorocyclohexane (β-HCH).

Lipid extraction

The analytical method is based on principles of liquid/liquid extraction. It was first described by Brevik (1978) and later modified by Polder et al. (2008). As the samples were also analysed for OH-metabolites, the modifications of Gabrielsen et al. (2011) were used. The procedure and results from these OH-analyses will not be discussed further in this study. After manual homogenisation with a scalpel, 1g (±0.100g) of sample was weighed to three decimals and added internal standards (I.S.) of PCBs (PCB -29, -112, -207), 2mL 6% NaCl and 10mL 1M H_2SO_4. The first extraction was performed with acetone and cyclohexane (3:4). The samples were homogenised with Ultra Turrax (IKA Ultra-Turrax T25, IKA Laboratory

Technology, Staufen, Germany), prior to sonication with an ultrasonic homogeniser (Cole Parmer CPX 750, Vernon Hills IL, USA). After centrifugation (Allegra X-12R Beckman Coulter, Fullerton, CA, USA), the supernatants were collected. The extraction was repeated with a second acetone and cyclohexane (1:2) mixture, sonication for 1 minute and centrifugation. The lipid extracts were collected and evaporated down to 1mL at 40°C by a steady flow of nitrogen (purity: 99.6%; AGA AS, Oslo, Norway, pressure 0.6 bar) in a Zymark® evaporation system (TurboWap II, Zymark Corporation, Hopkinton, MA, USA). Finally, the concentrated lipid extracts were quantitatively transferred to 5mL volumetric flasks and adjusted to the respected volume with cyclohexane.

Gravimetric lipid determination and acid cleanup

Lipid determination was done gravimetrically using 1mL aliquot of the sample (Formula 1). The remaining 4mL lipid extract was cleaned twice by an excess amount of concentrated sulfuric acid, H_2SO_4 (purity 97.5%; Fluka analytical, Sigma-Aldrich, St.Louis, USA). The lipid-free extracts were evaporated by a flow of nitrogen to a final volume of 0.2mL. The final gas chromatography (GC) extracts were transferred to GC vials.

Formula 1

$$\frac{(\mathit{weight\ lipid\ of\ one\ mL\ lipid\ extract}) \times 5 \times 100}{\mathit{total\ weight\ sample}} = \mathit{fat\ percentage}$$

GC analysis of OCPs and PCBs

Details of the GC analyses of OCPs and PCBs are as described by Polder et al. (2008), with the following modifications in specifications of GC conditions: constant flow of hydrogen carrier gas was increased to 1.2mL/min and final holding time at 275°C was increased to 21 min, making the total run time 76.6 min. The lowest level of detection (LOD) for individual compounds was determined as three times the noise level, except for *p,p'*-DDT, *cis*-chlordane, PCB-28, -52, and -101. Due to co-eluting compounds the limit of quantification (LOQ) for these compounds was set to ten times the noise level. None of these three respective compounds were detected in concentrations higher than LOQ in any of the analyzed samples (n=100). These compounds were therefore excluded from further statistical evaluations. The LODs/LOQs were: PCBs: 0.186 – 3.439 ng/g wet weight (w.w.), *cis*-chlordane: 1.861 ng/g w.w., *trans*-nonachlor: 0.202, oxychlordane: 0.188, *p,p'*-DDT: 4.490

ng/g w.w., *p,p′*-DDE: 0.349 ng/g w.w., HCB: 0.125 ng/g w.w., mirex: 0.192 ng/g w.w. and β-HCH: 0.489 ng/g w.w.

Quality control

For each series of 15 samples, 3 blank samples, one blind and 2-3 recovery samples were run, as well as the in-house reference material of seal blubber. The blank samples consisted of internal standards (I.S.) and solvents. Positive blanks were observed for some of the compounds. The mean procedural blanks were subtracted for these compounds. Recoveries ranged between 88 and 118% for OCPs and between 107 and 128% for PCBs. Results were not corrected for recovery. Other quality parameters were found satisfactory.

2.6 Data analysis

Statistical analyses included POPs detected in more than 70% of the samples (Table 1). Values below the LOD were replaced by half the detection limit before lipid normalization of the data. Statistical analyses were carried out using the statistical program R, version 3.0.1 (R core team, 2013). Principal component analysis (PCA) was performed to explore the relationship between different POPs (Quinn and Keough, 2011). As this analysis indicated high correlations between POPs (Appendix 1), the contaminants were grouped after their chemical properties. This was done to reduce the number of statistical tests, as well as to simplify the interpretation and presentation of the results. The contaminants were divided into three groups: group 1: ∑PCBs (PCB -28, -52, -101, -118, -138, -153 and -180), group 2: ∑chlordanes (*cis*-chlordane, trans-nonachlor and oxychlordane) and group 3: *p-p′*-DDT, *p,p′*-DDE, β-HCH, mirex and HCB (Table 1). Although the chlordanes in group 2 are pesticides, the group 3 compounds will be referred to as organochlorinated pesticides (OCPs).

A strong correlation between δ^{13}C and δ^{15}N was observed, which prevented the inclusion of both SIA values as explanatory variables in the statistical analysis (Appendix 2). We chose to focus on the stable isotopes of carbon (δ^{13}C) in our linear models, as δ^{13}C is more suitable to separate between marine and terrestrial diets (Kelly, 2000). The explanatory variables δ^{13}C, reindeer mortality and sea ice index were centred and scaled before analysis to facilitate the comparison of effect size (Gelman and Hill, 2007). All contaminant data were ln-transformed and diagnostic plots were used to determine if the data met the assumptions of the parametric tests applied. Specifically, the assumptions of constant variance and approximate normal distribution of residuals were assessed through plots of residuals against fitted values and normal quantile-quantile plots (Zuur et al., 2010). One individual with extreme values of stable isotopes was identified and excluded from the statistical analyses. The removal of this individual had no effect on the final results of the statistical analyses (not shown). There was a large difference in arithmetic and geometric mean of *p,-p′*-DDE due to one individual with extreme concentrations of this contaminant (table 1). However, this individual also had the highest concentrations of mirex, PCB-118, PCB-153, and ∑POPs. As the removal of this individual did not change the results of our statistical analyses (not shown), we chose to keep this outlier in our data for statistical analyses.

Linear models (LR) and Akaike's Information Criterion (Burnham and Anderson, 2002) corrected for small sample size (AIC_c) were used to investigate POP concentrations against sex, age, body condition, year, $\delta^{13}C$, reindeer mortality and sea ice coverage. More specifically, AIC_c was applied to a list of candidate models to find the most parsimonious model explaining the response. In addition to a specific AIC_c value, we calculated the number of parameters (K), the difference in AIC_c values between the "best" model and the model at hand (ΔAIC_c) and a normalised weight of evidence in favour of the specific model, relative to the whole set of candidate models, derived by $e^{(-0.5(\Delta AICc))}$ (AICc weights). Generally, models with ΔAIC_c values < 2 are relatively well supported by the data, whereas values between 3 and 7 indicate low relative support for the model, and models with values ΔAIC_c >10 can be seen as being unsupported (Burnham and Anderson, 2002). As several of our models had substantial support ($\Delta AIC_c < 2$), model averaging was used to make inference from all models. This method produces averaged estimates of all explanatory variables in the candidate list, weighted using the AIC_c weights (Burnham and Anderson, 2002; Lukacs et al., 2010). As confidence intervals (CI) are intervals covering the true value of the parameter 95% of the time, we considered confidence intervals of the model averaged estimates excluding zero as showing a statistically significant effect (i.e., for which there is strong evidence for an effect). Model-averaged confidence intervals have much better coverage properties than post-selection confidence intervals (i.e., ignoring the model selection process; Burnham and Anderson (2002)).

3. Results

3.1 POP concentrations

Table 1 shows the concentrations of POPs in liver tissue of arctic foxes from Svalbard averaged over all years, as well as the structural organisation of the different groups. In general, contaminant concentrations were highly variable and influenced by individuals with high concentrations. Both ∑PCBs and ∑chlordanes made up over 45% of ∑POPs, thereby dominating the quantitative amount of all POPs analysed. For ∑PCBs, only the higher chlorinated congeners (PCB -118, -138 -153 and -180) were detected. The most prevalent PCB was PCB-180, followed by PCB-153 and PCB-138. The composition and levels of chlordane related compounds (∑chlordanes) were highly variable. This group constituted mainly of the chlordane metabolite oxychlordane, detected in all samples and contributing 46.5% to the total POP concentrations. *Cis*-chlordane was not detected in any of the samples and trans-nonachlor was detected in 89% of the samples, but at low concentrations.

Table 1: Levels of POPs (ng/g l.w.) in liver tissue of arctic foxes from Svalbard, Norway in all study years (n=100). SD = standard deviation, min = minimum value, max= maximum value, %∑POPs = percentage of total POPs analysed

analytes	n	arithmetic mean	geometric mean	SD	min	max	% ∑POPs
lipid %	100	7.4	6.9	3.2	3.6	18	
PCB-28		nd	nd	nd	nd	nd	0.0
PCB-52		nd	nd	nd	nd	nd	0.0
PCB-101		nd	nd	nd	nd	nd	0.0
PCB-118	96	247	102	705	5.1	6 840	1.0
PCB-138	98	707	271	1 268	11	8 401	2.9
PCB-153	100	4 528	2 424	5 138	69	25 005	19.0
PCB-180	99	5 977	2 982	5 955	67	28 045	24.8
∑PCBs		11 378	6 121	11 962	69	53 129	47.6
cis-chlordane	0	nd	nd	nd	nd	nd	0.0
trans-nonachlor	89	440	43	1 293	2.4	7 486	1.6
oxychlordane	100	11 114	6 068	10 632	157	48 459	46.5
∑chlordanes		11 503	6 229	11 025	157	48 722	48.2
DDT	0	nd	nd	nd	nd	nd	0.0
p,-p′-DDE	73	232	20	1 065	1.6	9 099	0.7
HCB	100	201	129	208	8.8	1 030	0.8
Mirex	100	575	333	619	13	3 941	2.4
β-HCH	92	62	34	79	3.1	527	0.2
∑OCPs		1 006	593	1 534	31	14 156	4.2

Although *p,p′*-DDE, HCB, mirex and β-HCH (∑OCPs) were present in the majority of the samples, they were of quantitatively little importance, making up less than 5% of the total POP concentrations. *p-p′*-DDT was not detected in any of the samples. However, the DDT metabolite, *p,p′*-DDE was detected in small amounts in 73% of the samples. The most predominant OCP was mirex, present in all analysed samples and constituting 2.4% of ∑POPs analysed.

3.2 Factors underlying contaminant trends

The results of our highest ranked models (ΔAICc < 2) and the model averaged parameters are presented in Table 2. Body condition, year and $\delta^{13}C$ were present in nearly all of the highest ranked models (i.e., very high importance). The model averaged parameters for the effect of these variables had similar estimates as those presented in the highest ranked models. There was a significant effect of increased body condition and $\delta^{13}C$ on POP concentrations. There was also a temporal decrease in POP concentrations over the 13 year period of the study. ∑PCBs and ∑chlordanes decreased -10.4% [95% CI: -15.6, -4.9] and -7.7% [95% CI: -13.1, -1.0] per year, respectively. ∑OCPs showed a non-significant decreasing trend of -3.9% [95% CI: -8.6, 1.0] per year. These values were calculated using the model averaged estimates for year, and are thus corrected for potentially confounding factors such as sex, age, body condition and diet.

Reindeer mortality had a non-significant negative effect on all three groups of contaminants. Sea ice coverage during the ringed seal pupping period had diverse and non-significant effects on the three groups of contaminants. Sex and age were not present in any of the highest ranked models. In addition, model averaging of these variables found no trend in their estimates and a large spread in the CI, illustrating that there was no evidence for an effect of sex and age on the levels of POPs in our data.

Table 2: Highest ranking models from AICc (Δ AIC < 2) and model averaged results (n=99) explaining the concentrations of ∑PCBs (PCB -118, -138, -153, -180), ∑chlordanes (trans-nonachlor, oxychlordane) and ∑OCPs (*p-p*-DDE, HCB, mirex, β-HCH) in liver of arctic foxes from Svalbard

	model parameters								AIC		
model rank	intercept	sex	age	body condition	year	$\delta^{13}C$	reindeer mortality	sea ice coverage	K	Δ AIC	AIC weights
Group1; PCB 118, -138, -153, -180											
1	10.1167			-0.3660	-0.1077	0.5690			5	0.00	0.36
2	10.0784			-0.3685	-0.0977	0.5580	-0.0980		6	1.33	0.19
3	10.1764			-0.3650	-0.1215	0.5593	-0.1883	-0.1976	7	1.78	0.15
4	10.1658			-0.3638	-0.1199	0.5732		-0.0729	6	1.91	0.14
model average	10.10	0	0.2	-0.37	-0.11	0.56	-0.15	-0.16			
95 % CI	9.50 , 10.71	-0.38 , 0.38	-0.17 , 0.58	-0.55 , -0.18	-0.17 , -0.05	0.37 , 0.76	-0.40 , 0.09	-0.46 , 0.14			
Group 2; trans-nonachlor, oxychlordane											
1	10.0329			-0.3697	-0.0821	0.5138			5	0.00	0.29
2	9.9680			-0.3741	-0.0652	0.4952	-0.1661		6	0.09	0.27
3	10.0281			-0.3719	-0.0799	0.4960	-0.2215	-0.1212	7	1.84	0.11
model average	9.96	-0.15	-0.05	-0.37	-0.08	0.51	-0.2	-0.04			
95 % CI	9.29 , 10.64	-0.57 , 0.27	-0.47 , 0.37	-0.57 , -0.17	-0.14 , -0.01	0.29 , 0.72	-0.45 , 0.05	-0.41 , 0.33			
Group 3; *p-p*-DDE, HCB, mirex, β-HCH											
1	7.1562			-0.2429	-0.0360	0.5208	-0.1364		6	0.00	0.20
2	7.2094			-0.2393	-0.0499	0.5361			5	0.12	0.19
3	6.9866			-0.2395		0.5361	-0.1913		5	0.23	0.18
4	6.9906			-0.2411		0.5410		0.1728	5	1.23	0.11
5	7.1537			-0.2417	-0.0361	0.5313		0.0825	6	1.79	0.08
6	6.9928			-0.2420		0.5312	-0.1365	0.0854	6	1.85	0.08
model average	7.09	0.07	0	-0.24	-0.04	0.53	-0.15	0.08			
95 % CI	6.56 , 7.61	-0.27 , 0.4	-0.33 , 0.33	-0.40 , -0.08	-0.09 , 0.01	0.36 , 0.70	-0.35 , 0.04	-0.17 , 0.34			

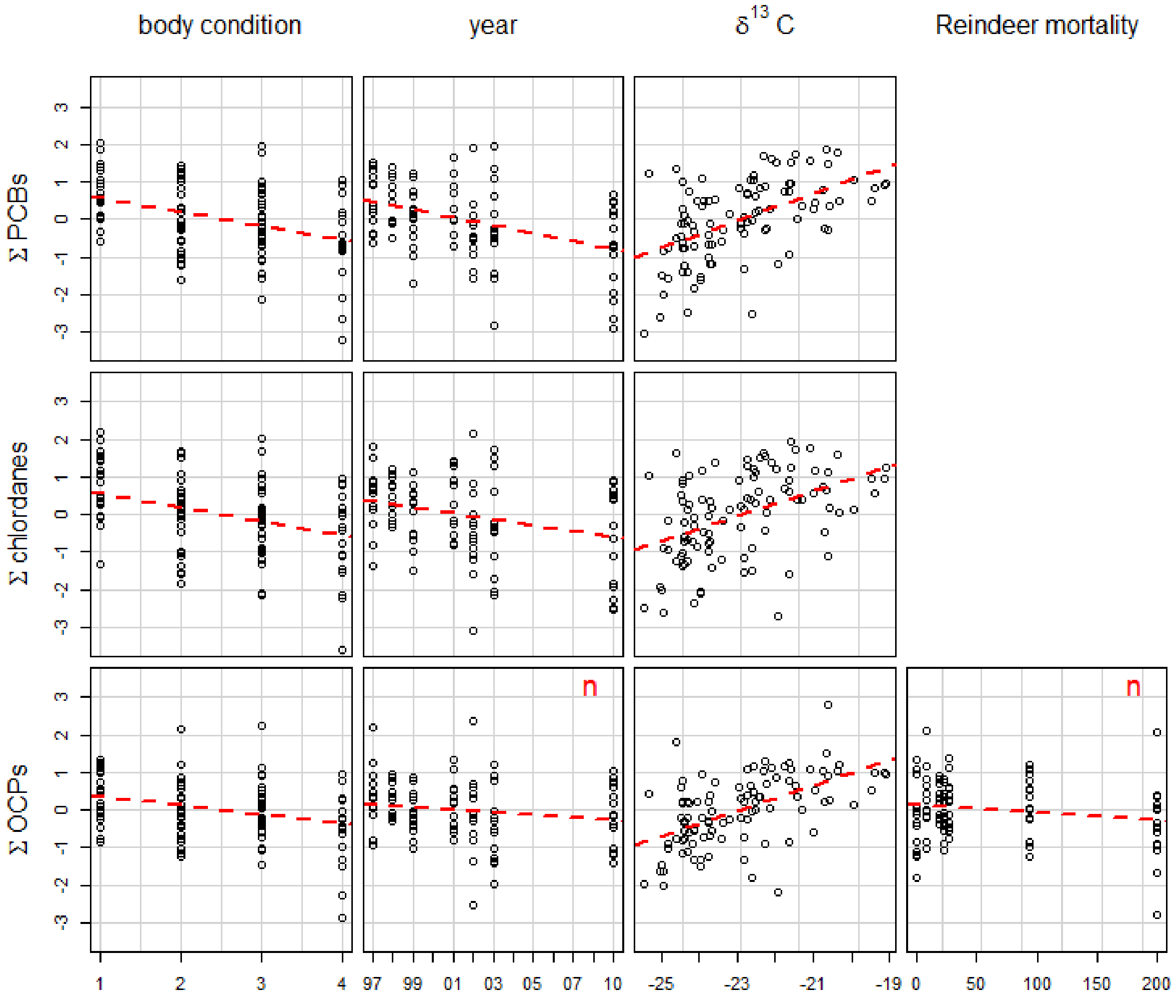

Figure 2: Relationships between the concentrations of ∑PCBs (PCB -118, -138, -153, -180), ∑chlordanes (trans-nonachlor, oxychlordane) and ∑OCPs (*p-p'*-DDE, HCB, mirex, β-HCH) in liver of arctic foxes from Svalbard and the predictor variables: body condition, year, δ^{13}C and reindeer mortality. The plots are partial residuals plots based on the highest ranked models in table 2 (i.e. the effects of each predictor variable have been controlled for the other variables in the model). Note that the partial residuals are ln-transformed and the variables δ^{13}C and reindeer mortality are presented in their original conformation. For the variable year; each season is represented by the year starting the trapping season (e.g. 1997/98="97", 1999/00="99" etc.). Red "n" mean no evidence of effect (i.e. confidence intervals not excluding zero).

4. Discussion

4.1 POP concentrations

Several previously published papers on POPs in arctic foxes have used w.w. to report their results. All values presented and discussed in this paper are in l.w., although w.w. concentrations were used for comparison with previous research. Comparisons between POP concentrations measured in different tissues of foxes in this study were done using l.w. concentrations. Furthermore, geometric means were preferred over arithmetic means whenever possible.

In this study, the levels of ∑PCBs were present at relatively high concentrations, with a large spread in the contaminant concentrations, both between individuals and between years (Table 1, Appendix 3). In fact, the within year concentrations of ∑PCBs in the present study more or less covers the range of all previous ∑PCB measurements in arctic foxes from Svalbard (Appendix 3). For further discussion, we will use the yearly average of contaminant concentrations presented in Table 1. Compared with previous studies on levels of POPs in arctic fox liver from Svalbard, concentrations of ∑PCBs in the present foxes were about half of those found in foxes from 1991/92 (Severinsen and Skaare, 1997), and within the same range as in foxes collected in 1973/74 and 1983/84 (Norheim, 1978; Wang-Andersen et al., 1993). The ∑PCB concentrations in the present study were more than seven times higher than ∑PCB concentrations found in the liver of arctic foxes captured in 1999/00 from Canada and Alaska (Hoekstra et al., 2003a).

The levels of ∑chlordanes in the present study were within the same range as those measured in the liver of arctic foxes collected on Svalbard in 1991/92 (Severinsen and Skaare, 1997). In the present study, levels of the metabolite oxychlordane were far above the levels of trans-nonachlor and cis-chlordane, both found in technical chlordane (Karlsson et al. (1999). Oxychlordane has been reported to biomagnify in the food web (Hoekstra et al., 2003b), and high levels of this POP relative to other chlordanes have previously been reported in numerous studies of seabirds and marine mammals in the Barents Sea (Bustnes et al., 2010; Gabrielsen et al., 1995; Routti et al., 2009; Sagerup et al., 2009). However, the higher ratio of oxychlordane to ∑chlordanes in the present foxes compared to marine mammals from the same region (Andersen et al., 2001; Routti et al., 2009) provides further support to the hypothesis that arctic foxes are capable of metabolizing this compound class (Fuglei et al., 2007; Hoekstra et al., 2003a).

In the present study, oxychlordane was the major POP detected. This is in accordance with results previously found in liver of farmed arctic foxes fed with a contaminated diet of minke whale (*Balaenoptera acutorostrata*) blubber and wild arctic foxes from Canada and Alaska (Helgason et al., 2013; Hoekstra et al., 2003a). The concentrations of oxychlordane in the present foxes were four times higher than those previously reported in arctic fox liver from Canada and Alaska (in 1999/00) and four times lower than those found in liver of four coastal arctic foxes collected on Iceland in 1993/94 (Hoekstra et al., 2003a; Klobes et al., 1998). In addition, the levels of oxychlordane in liver of the present foxes were more than two times higher than the levels measured in adipose tissue of arctic foxes from Svalbard (Fuglei et al., 2007). However, a recent study on farmed arctic foxes fed with a contaminated diet reported high concentrations of chlordanes in hepatic tissue compared to other organs and relative to ∑PCB concentrations (Helgason et al., 2013). The authors argued that this may be due to hepatic sequestration of oxychlordane through inductions of xenobiotic-metabolizing enzymes, as previously found in raccoon dogs (Kunisue et al., 2007).

None of the 100 foxes analysed in this study had levels of *p-p′*-DDT above the LOQ. The only previous measurement of DDT in liver of arctic foxes from Svalbard reported levels above 600ng/g (Severinsen and Skaare, 1997). Hence, there seems to have been a dramatic decline of DDT in this population. This apparent reduction of DDT is in accordance with significant declines of DDT reported elsewhere in arctic biota (Rigét et al., 2010). The DDT metabolite, *p,p′*-DDE was present above the limit of detection in 74% of the samples. The only previous measurement of this compound in arctic foxes from Svalbard was measured in adipose tissues of foxes collected in 1998/99 (Fuglei et al., 2007). The liver concentrations of *p,p′*-DDE in the present study were more than 50% lower than the results presented by Fuglei et al. (2007). The concentrations of *p,-p′*-DDE presented in this study were about 10 times higher than those previously found in liver of foxes from Canada, Alaska and foxes with a terrestrial diet on Iceland (Hoekstra et al., 2003a; Klobes et al., 1998).

In the present study, HCB was found in all analyzed liver samples. The levels were about 50% lower than those previously reported in the liver of Svalbard arctic foxes (Severinsen and Skaare, 1997) and is in the same range as levels found in adipose tissue of foxes captured in 1998/99 from Svalbard (Fuglei et al., 2007). In comparison with previous studies elsewhere in the Arctic, our results of HCB were about five times higher than those found in the liver of foxes captured in Canada in 2000 (Hoekstra et al., 2003a), and within the same range as liver concentrations in foxes captured in 1993/94 and 1999 from Iceland and Alaska (Hoekstra et al., 2003a; Klobes et al., 1998). The concentration of β-HCH was the lowest of all detected POPs in this study. Our results of β-HCH were less than half of the previously reported concentrations in liver of arctic foxes from Svalbard (Severinsen and Skaare, 1997). The concentrations of β-HCH were within the same range as levels previously found in liver of foxes captured on Iceland in 1993/94, and in Canada in 2000 (Hoekstra et al., 2003a; Klobes et al., 1998). β-HCH concentrations were about one fourth of the previously reported concentrations in liver of foxes from Alaska (Hoekstra et al., 2003a).

Mirex was the most dominant OCP in the present study. To our knowledge, the only previously reported results of mirex in arctic foxes are from Severinsen and Skaare (1997), measured in liver of foxes from Svalbard. The results presented in this study were close to their findings and, as we do not know the number of foxes analyzed or the gender and age distribution in their study, the results were hard to compare.

With the exception of the lower chlorinated PCB congeners and β-HCH, all POPs were found at similar or higher concentrations compared to previous reports of POPs in liver of arctic foxes from Canada and Alaska. These results were in accordance with the observed spatial trends of POPs in other arctic apex predators, with increasing levels of the heavier POPs in eastern populations of polar bears (Letcher et al., 2010; McKinney et al., 2011; Verreault et al., 2005). However, the difference in POP concentrations between the different arctic fox populations within the arctic may also be attributed to the more marine diet in the "costal ecotype" arctic foxes from Svalbard, compared to the "lemming ecotype" arctic foxes collected in Canada and Alaska, as suggested in another study (Fuglei et al., 2007).

4.2 Factors underlying contaminant trends

The most important predictors explaining the variation of POP concentrations were body condition, year and $\delta^{13}C$. The estimates of these predictors did not change much within the highest ranked models of each contaminant group (Table 2). Therefore, the best model for each contaminant group was used to illustrate the effect of the predictor variables (Figure 2).

In the present study, we found strong evidence that higher POP concentrations are related to larger $\delta^{13}C$ values, which reflects increasing marine diet. In our results, $\delta^{13}C$ values ranged from -25.4‰ to -19.1‰ (Appendix 2). The expected mean $\delta^{13}C$ for a terrestrial herbivore is about –25‰ (Roth, 2002) and between -19 and -17‰ for a marine mammal (Hindell et al., 2012; Hoekstra et al., 2003b). As the $\delta^{13}C$ variable presented in Table 2 was centred and scaled, a one unit increase in $\delta^{13}C$ equals about 25% of the total range in the original $\delta^{13}C$ variable. We hypothesise that the evidence for low levels of POPs in foxes with a terrestrial diet reflects the minute POP concentrations in Svalbard reindeers (Gabrielsen et al., 2012; Polder et al., 2009; Severinsen and Skaare, 1997). Previous studies on polar bears foraging on the sea ice have found evidence for a significant relationship between $\delta^{13}C$ and POP concentrations (McKinney et al., 2013; McKinney et al., 2009). In the previous assessment of contaminants in Svalbard arctic foxes, Fuglei et al. (2007) found significant effects of increased trophic position, measured through $\delta^{15}N$ on POP concentrations. We did not include $\delta^{15}N$ in our statistical analysis. However, we hypothesise that the significant effect for $\delta^{13}C$ on variation in POP concentrations may also reflect a difference in trophic level ($\delta^{15}N$) of the sampled foxes.

Our results strongly suggest that body condition can affect concentrations of POPs in arctic foxes, as higher levels of contaminants were found in lean than fat foxes. This is in accordance with previous results in Svalbard arctic foxes (Fuglei et al., 2007; Wang-Andersen et al., 1993). The increased liver POP concentration in lean foxes is likely to originate from a remobilisation of the lipophilic POPs, where POPs are transported by the blood supply from adipose to hepatic tissues (Helgason et al., 2013). As arctic foxes undergo large seasonal variations in lipid reserves (Prestrud and Nilssen, 1992), the redistribution of contaminants is hypothesised to be severe in this species (Fuglei et al., 2007).

To our knowledge, this study is the first to investigate temporal trends of POPs in arctic foxes from Svalbard. The temporal decline of POPs was significant for ∑PCBs and ∑chlordanes, albeit not for ∑OCPs. The lack of evidence for a decrease of ∑OCPs (*p,p'*-DDE, HCB, mirex and β-HCH) differs from what is found elsewhere in arctic biota, where significant declining trends have been reported for these POPs (Dietz et al., 2013; Rigét et al., 2010). However, several of these trends were not adjusted for body condition or diet and diet, both of which were shown to have large effects in this study. The missing evidence for a decline in ∑OCPs in the present foxes might also be due to the nature of our data, with relatively few study years and many foxes with ∑OCP concentrations close to the LOD.

There was no evidence of a significant effect of sex and age on POP concentrations in the present study. The lack of evidence for these variables was expected, as the females selected for this study were too young to have given birth. Differences in POP concentrations between sexes are often observed in mammals, as mothers can pass on the lipophilic POPs to their offspring, first through the placenta barrier and later through the breast milk (Debier et al., 2003; Norstrom et al., 1998). The missing evidence of effects of sex and age presented in this study is in accordance with previous studies performed on similar data of arctic foxes from Svalbard (Fuglei et al., 2007; Wang-Andersen et al., 1993).

The evidence for an effect of reindeer mortality on POP concentrations was weak, with model averaged estimates of the CIs, presented in Table 2, overlapping zero. Despite the negligible levels of contaminants in Svalbard reindeers (Gabrielsen et al., 2012; Polder et al., 2009; Severinsen and Skaare, 1997), our results show that the mortality level of Svalbard reindeers has no statistically significant effects on POP concentrations in arctic foxes from Svalbard. However, the results showed a consistent trend, with lower levels of POPs associated with increasing number of reindeer carcasses. Such a trend may indicate biological significance (Yoccoz, 1991) of reindeer carcasses. The reindeer mortality data were scaled prior to statistical analyses, and a one unit increase in the scaled reindeer mortality equals an increase of approximately 66 reindeer carcasses. As illustrated in Figure 2, mortalities can be much higher than this. Indeed, in earlier seasons the number of reindeer carcasses has reached levels well above 200 individuals in the Adventdalen population alone (Tyler and Øritsland, 1999). We therefore will not exclude the possibility that years with extreme mortality levels of Svalbard reindeers may affect the POP concentrations in arctic foxes from Spitsbergen.

There was no consistent effect of sea ice cover on POP concentrations in the present study. Due to the nature of the raw data of the sea ice variable (m^2 sea ice between 15th March and 30th April) it is more convenient to present this variable in the scaled configuration. The data include seasons with low and high ice cover (Pavlov et al., 2013; Zhuravskiy et al., 2012). The missing evidence of an effect of sea ice cover on POP concentrations in arctic foxes might be due to the large uncertainties in this variable. We used total area of sea ice in Isfjorden, the largest of the west Spitsbergen fjords (Nilsen et al., 2008). Any calculations based on satellite measurements of visible ice in this area over a longer time period needs to be considered as rough estimates.

The low and ambiguous effect of sea ice cover on the levels of POPs in arctic foxes may also reflect that area of sea ice cover is not the single variable influencing arctic fox hunting conditions on the sea ice. Seal pups are born in a subnivean birth lair, dug out from snow accumulated on the sea ice. In years with low snow cover, ringed seal pups can be born without cover on the surface of the ice, but the mortality rate of these pups is thought to be extremely high (Kovacs and Lydersen, 2006; Smith et al., 1991). Hence, other variables, such as precipitation and snow cover, might be equally important. Several previous studies have suggested that sea ice might have an effect on the arctic fox population on Svalbard (Eide et al., 2012; Frafjord, 1993; Prestrud, 1992), but few have actually included ice conditions in their studies. To our knowledge, no study has previously investigated the effects of sea ice on POP concentrations in arctic foxes. For further investigations, we suggest to include snow cover, either as an independent variable or through interactions with sea ice coverage.

4.3 Relationship between $\delta^{13}C$, reindeer mortality and sea ice cover

Linear regressions revealed a weak and insignificant relationship between $\delta^{13}C$ and reindeer mortality ($R^2 = 0.02$, $p > 0.05$; not shown). This was also the case for $\delta^{13}C$ and sea ice cover ($R^2 = 0.02$, $p > 0.05$; not shown). We expect the SIA to cover dietary inputs two months back in time, and both reindeer mortality and sea ice cover describe possible dietary opportunities in spring preceding the trapping season. As the foxes were captured in November – March, the time interval between validity of the SIA and the climatic variables was probably too large to have an effect. In addition, factors not considered in this study, such as the occurrence of migratory sea birds, might impact the arctic fox diet (Angerbjorn et al., 2004; Eide et al., 2012; Prestrud, 1992).

The weak evidence for effects of reindeer mortality and sea ice cover on levels of POPs differs from the results of previous studies investigating effects of ecological parameters on POP concentrations in biota. A recent study reported snow depth and vole abundance as critical factors explaining the variance in PCBs and DDE in snowy owls (*Strix aluco*) from Central Norway (Bustnes et al., 2011). Similarly, Gaden et al. (2012) reported that the sea ice break up date had a significant effect on concentrations of PCBs and DDE in ringed seals from western Canada. A possible explanation for the weak evidence of effects for reindeer mortality and sea ice coverage on POP concentrations in the present study might be the existing correlations between year, reindeer mortality and sea ice cover. Appendix 4 shows the relationships between the three variables, with the scaled reindeer mortality and sea ice cover plotted against year. From this, it seems evident that one might expect high sea ice coverage in years with low reindeer mortality. This relationship was also apparent in the results of the highest ranked models, presented in Table 2. The estimates of the three predictor variables changed when one or both of the other two variables were present. These changes were particularly evident in the ∑OCP models, where the effect of sea ice increased by a factor of two, depending on whether year or reindeer mortality was included in the model. Reindeer mortality also changed by a factor of two in the ∑PCB models, dependent on the presence of sea ice coverage.

4.4 Influence of climate change on POP concentrations in arctic foxes from Svalbard

In the Arctic, a warmer climate is expected to increase the frequencies of winters with "rain-on-snow" events (Putkonen and Roe, 2003; Rennert et al., 2009). Recent studies have shown that this will lead to higher numbers of Svalbard reindeer carcasses in winter (Hansen et al., 2011; Hansen et al., 2013). Despite the weak evidence of effects for reindeer mortality on POP concentrations in arctic foxes, we hypothesise that more frequent years with extreme reindeer mortalities may contribute to a decrease in the arctic fox POP concentrations.

A warmer climate is also expected to decrease the sea ice cover of Svalbard fjords (Pavlov et al., 2013; Zhuravskiy et al., 2012). Any possible effects of this decrease in sea ice cover on POP concentrations in arctic foxes remains uncertain, as our results showed weak and ambiguous effects of sea ice coverage in Isfjorden. As previously mentioned, we suggest precipitation and snow cover as possible factors determining arctic fox hunting conditions on the sea ice. A warmer climate is also expected to decrease snow cover in the Arctic (Liston and Hiemstra, 2011), and increase the amount of rain on Svalbard (Førland et al., 2011; Førland and Hanssen-Bauer, 2003). More rain may decrease ringed seal pup survival and facilitate the arctic fox hunting conditions on ringed seal pups, as rain can destroy the structure of the ringed seal subnivean birth lairs (Stirling and Smith, 2004).

In terms of POP concentrations, the strong effects of diet in the present study suggest that any climatic events with sufficient ecological impact to induce a shift in the diet of the arctic foxes are likely to influence the POP concentrations in the arctic fox population on Svalbard.

4.5 Conclusion and future perspectives

We found evidence of a temporal decrease in contaminants over the 13 year period of the study. There was a significant decline in ∑PCBs and ∑chlordanes and a non-significant decrease in ∑OCPs, when controlling for confounding factors such as body condition and diet. Although our data does not represent an optimal time trend series, our results illustrate the importance of seeking robust time trends of POPs in biota and adjusting these for possible confounding factors whenever possible. We also found significant evidence for negative effects of increasing body condition and a terrestrial diet on POP concentrations. Reindeer mortality had non-significant negative effects in all three groups of contaminants, while sea ice coverage had weak and ambiguous effects on the POP concentrations. Although it is highly likely that arctic foxes are affected by the between year climatic variation on Svalbard, connecting the levels of POPs in arctic foxes with reindeer mortality and sea ice cover has proved challenging.

For future research, we suggest to increase the sample size by incorporating more years in the data set. This would enable a more accurate picture of the contaminant concentrations in the population and possibly exclude the co-linearity observed between year, reindeer mortality and sea ice cover. We also suggest an improvement in the determination of the sea ice cover, as we currently believe the lack of evidence in this variable may be due to its inherent uncertainties.

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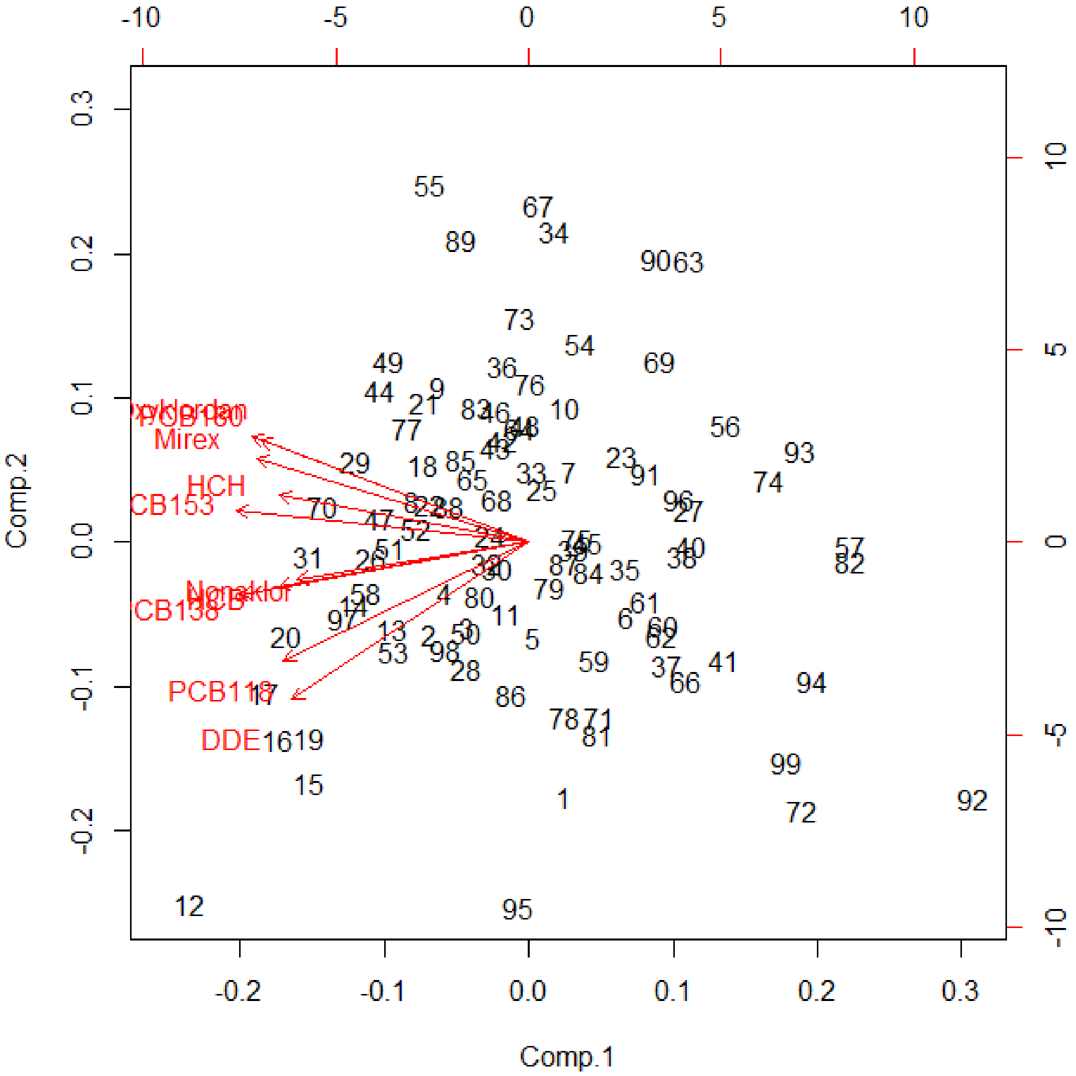
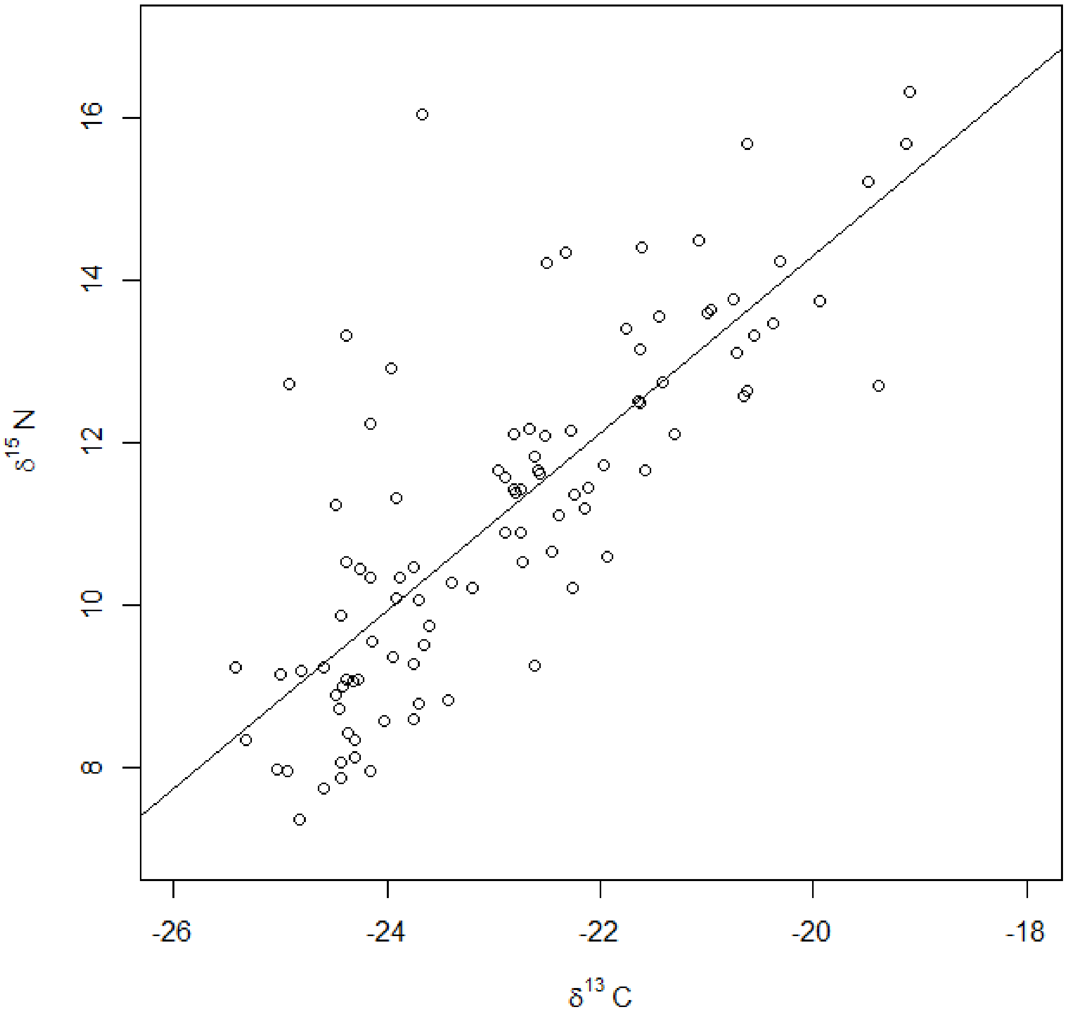
Appendix 1



Appendix 1: Plot of principle component analysis (PCA). Principle component 1 explained 73% of the variance and principle component 2 explained 8%. All contaminant data was log transformed prior to analysis.

Appendix 2



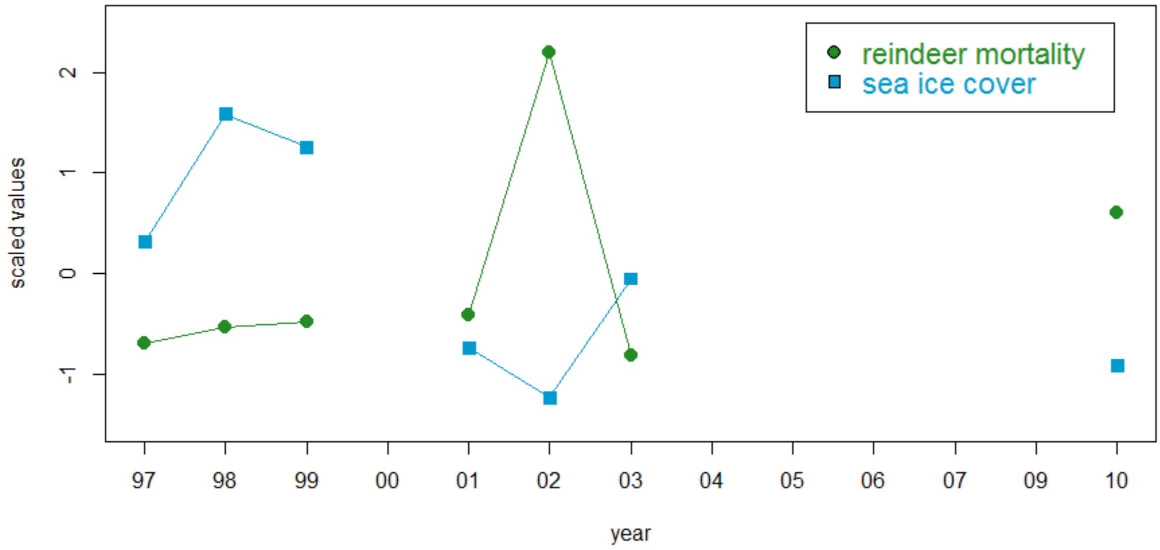
Appendix 2: Linear regression illustrating the relationship between δ^{15}N and δ^{13}C ($F_{1, 158}$, $p<0.0001$. $R^2= 0.62$).

Appendix 3

Appendix 3: Levels of POPs (ng/g l.w.) in liver tissue between years of arctic foxes from Svalbard, Norway (n=100), SD = standard deviation, min = minimum value, max= maximum value.

season	POPs	arithmetic mean	geometric mean	SD	min	max
1997 -1998	∑PCBs	14 582	9 823	13 740	2 492	5 3129
(n=14)	∑chlordanes	12 991	9 158	9 997	1 319	33 992
	∑OCPs	1 743	822	3 609	230	14 156
1998 - 1999	∑PCBs	19 008	12 681	14 549	1 244	46 959
(n=14)	∑chlordanes	17 095	12 692	11 284	1 706	36 504
	∑OCPs	1 531	1 129	1 071	223	3 616
1999 - 2000	∑PCBs	10 385	5 728	11 917	563	40 036
(n=14)	∑chlordanes	9 406	5 729	10 243	1 096	36 945
	∑OCPs	788	547	790	149	3 156
2001 - 2002	∑PCBs	16 812	13 847	10 763	5 131	34 988
(n=14)	∑chlordanes	16 595	13 610	10 274	4 422	37 232
	∑OCPs	1 164	1 011	619	332	2 608
2002 - 2003	∑PCBs	5 149	3 748	4 750	1 024	18 556
(n=14)	∑chlordanes	6 457	3 242	7 492	227	25 592
	∑OCPs	543	318	618	32	2 313
2003 - 2004	∑PCBs	9 363	4 292	12 800	233	49 656
(n=14)	∑chlordanes	10 740	4 219	15 009	333	48 722
	∑OCPs	644	381	719	60	2 541
2010 - 2011	∑PCBs	5 224	2 309	6 287	69	23 334
(n=16)	∑chlordanes	7 771	3 217	8 959	157	27 983
	∑OCPs	676	409	565	31	1 885

Appendix 4



Appendix 4: Number of reindeer carcasses and sea ice coverage in the study years. Reindeer mortality data is from Adventdalen. Sea ice cover is total m^2 fast ice for more than 14 days in Isfjorden during 15.March – 30.April. Both variables are centred and scaled.