Occurrence, bioaccumulation and long-range transport of short-chain chlorinated paraffins on the Fildes Peninsula at King George Island, Antarctica

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ABSTRACT

As a candidate persistent organic pollutant of the Stockholm Convention, short-chain chlorinated paraffins (SCCPs) have recently received particular attention. In this study, we investigated, for the first time, the concentrations of SCCPs in biota samples collected from the Fildes Peninsula at King George Island and Ardley Island, Antarctica. The concentrations of SCCPs ranged from 3.5 to 256.6 ng/g (dry weight, dw), with a mean of 76.6 ± 61.8 ng/g dw, which was lower than those detected in mid- and low-latitude regions. The long-range transport behaviour of SCCPs was confirmed by both the detection of SCCPs in Antarctic remote areas and their special congener profiles. Short carbon chain (C_{10}) congeners predominated in the Antarctic samples, which accounted for 56.1% of the total SCCP contamination. Such enrichment of C_{10} congeners indicated the high potential for the long-range transport of shorter chain congeners. In addition, SCCPs tended to be enriched in the species with high lipid contents. The biomagnification potential of SCCPs was found between *Archeogastropoda* (Agas) and *Neogastropoda* (Ngas), and the biomagnification factors of shorter chain congeners of SCCPs were higher than that of the longer chain ones. Considering that the endemic species in polar regions may be sensitive and vulnerable to the adverse effects of environmental contaminants, more attention should be paid on the bioaccumulation and toxicological risks of SCCPs in polar environments.

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1. Introduction

Chlorinated paraffins (CPs) have 10–30 carbon atoms and chlorine content that ranges from 30% to 70% by weight (Bayen et al., 2006). The variety of carbon chain lengths and chlorine contents of CPs results in diverse properties and extensive applications, including lubricant additives, plasticizers in rubbers, metal-working fluids, flame retardants, paints and sealants (Campbell and McConnell, 1980). Short-chain chlorinated paraffins (SCCPs, C_{10} – C_{13}) can affect the liver, thyroid hormone system, and kidneys, e.g., by inducing hepatic enzymes and hyperactivity in the thyroid, which can lead to both acute and chronic toxic effects with long-term exposure (UNEP/POPS/POPRC.8/6, 2012). Because of their potential for environmental persistence (Iozza et al., 2008), toxicity (Geng et al., 2015), bioaccumulation (Fisk et al., 1996; Houde et al., 2008; Ma et al., 2014b; Zeng et al., 2011b) and long-range atmospheric transport (Ma et al., 2014c; Vulykh et al., 2007), SCCPs have received widespread public attention. SCCPs were even included in Annex E of the Stockholm Convention by the persistent organic pollutants (POPs) Review Committee as a POP candidate in 2008 (UNEP/POPS/POPRC.3/20, 2008) since these properties are quite similar to those of legacy

POPs (Poremski et al., 2001). However, with the scarcity of current monitoring data, including for the occurrence, capacity for accumulation, and long-range transport, the listing of SCCPs as POPs into the Stockholm Convention remains controversial (Wang et al., 2013).

SCCPs are primarily released into the environment from anthropogenic sources as a type of high production volume chemical and are detected in the different environmental matrices, including air, water, sediments and biota (Barber et al., 2005; Bayen et al., 2006; Coelhan, 2010; Fridén et al., 2011; Marvin et al., 2003; Thomas et al., 2006; Zeng et al., 2015). The SCCPs travel to polar regions in a three-step process: release, transport and deposition (Wania, 2003). The vapour pressure of SCCPs is very low and varies from 2.8×10^{-7} to 0.066 Pa (Feo et al., 2009), which make them easily volatilize into the atmosphere. Most pollutants migrate to higher latitudes in a series of relatively short jumps called the “grasshopper effect” (Wania and Mackay, 1996). SCCPs can also be transported through the atmosphere and ultimately deposited in remote areas such as the polar regions (Wania, 2003; Wania and Mackay, 1993). The detection of SCCPs in the polar regions would provide direct evidence for their long-range transport, but SCCPs have been observed in polar regions in only a few studies. The concentrations of SCCPs were studied in the biota and sediment samples from Arctic and sub-Arctic environments (Reth et al., 2006; Strid et al., 2013; Tomy et al., 2000; Tomy et al., 1999). For Antarctica, the only data for SCCPs are from the air samples collected on the Fildes Peninsula (Ma et al., 2014c).

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Antarctica is far from industrial regions and is less affected by global anthropogenic activities. Most species found in Antarctica are endemics and may be more sensitive and vulnerable to the adverse effects of persistent contaminants than those of the industrialized regions. Low levels of pollutants can have significant effects on local fragile ecosystems (Bargagli, 2008). Moreover, biomagnification through food chains can lead to relatively high contaminant concentrations in biota, which potentially threaten the health of wildlife at higher trophic levels. In several studies, SCCPs biomagnified through the food web in aquatic ecosystems (Ma et al., 2014b; Zeng et al., 2011b). As an additional factor, the temperature in Antarctica is lower than that in other regions, and based on previous studies, low temperatures may decrease the growth rate more than the uptake rate, which results in a net accumulation of POPs in the tissues of biota (Honkanen and Kukkonen, 2006; Muijs and Jonker, 2009). Therefore, relatively high levels of bioaccumulation are expected for Antarctic species.

In this study, we systematically collected soil, vegetation, and animal samples on the Fildes Peninsula at King George Island and Ardley Island, Antarctica. The primary purpose was to investigate the occurrence and congener profiles of SCCPs in Antarctic samples to demonstrate the long-range transport behaviour of SCCPs. The spatial distribution of SCCPs was also examined to determine the effect of human activities on the local Antarctic environment. Finally, the bioaccumulation behaviour of SCCPs was investigated to study the effects on the Antarctic ecosystem. The results of our study will provide valuable information on the environmental behaviour of SCCPs on a global scale.

2. Materials and methods

2.1. Sampling

The samples were all collected during the 29th Chinese Antarctic Expedition between December 2012 and January 2013. Eight soil, six algae (*Halymenia floresia*), six moss (*Sanionia uncinata*), three fish (*Notothenia*

coriiceps), five neogastropoda (*Trophon geversianus*, Ngas) and sixteen archeogastropoda samples (*Nacella concinna*, Agas) were collected on the Fildes Peninsula at King George Island and Ardley Island, Antarctica (Fig. 1). The samples were immediately freeze-dried and homogenized at the Chinese Great Wall Station and were then transported to the laboratory in Beijing, China, and preserved at -20°C in a specialized refrigerator for polar samples until analyses. Detailed information on sampling and transportation is in the Supplementary Data.

Of the sampled species, the species of Agas was the most widely distributed on the Fildes Peninsula and therefore was the sentinel species used to investigate the spatial distribution of SCCPs. Moreover, the predator-prey relationship was confirmed for Agas and Ngas (Ngas preys on Agas; Fig. S1), and therefore, paired samples of Agas and Ngas were collected from sites 5, 7, 9, 12 and 18 (Fig. 1) to study the biomagnification behaviour of SCCPs.

2.2. Analytical procedures

Approximately 5 g of soil and 2 g of biological samples were spiked with surrogate standards (1 ng of $^{13}\text{C}_{10}$ -trans-chlordane), mixed with 15 g of anhydrous sodium sulphate and extracted with dichloromethane and hexane (1:1, v/v) using accelerated solvent extraction (Dionex ASE 350). The extract was cleaned by a multilayer Silica-Florisil column that contained 3 g of Florisil, 2 g of activated silica gel, 5 g of acid silica gel (30%, w/w) and 4 g of anhydrous sodium sulphate from the bottom to the top. The cleanup procedure for the SCCPs was based on our previously reported methods (Zeng et al., 2011a).

The SCCPs were analysed on a 7890A high-resolution gas chromatograph (HRGC) in electron capture negative ion (ECNI) mode coupled with a 7000B triple quadrupole mass spectrometer in single quad mode (Agilent, USA). 1 μL of extract was injected with a 7683B Series Injector (Agilent, USA) in splitless mode into a DB-5MS capillary column (30-m length, 0.25-mm i.d., 0.25- μm film thickness; Agilent, CA).

Fig. 1. Distribution of the sample sites and the concentrations of SCCPs in Antarctic samples. Numbers 1–23 are the different locations; detailed information is provided in Table S1.

To ensure the validity of analytical identifications and quantifications, strict quality assurance and control measures were employed. The concentrations of SCCPs detected in anhydrous sodium sulphate (blank controls throughout the treatment process) were 0.67–1.27 ng/g, which was <5% of sample concentrations. The recovery of $^{13}\text{C}_{10}$ -trans-chlordane was between 76% and 102%. The recoveries for the three SCCP references with chlorine contents of 51.5%, 55.5%, and 63.0% were between 81% and 116%. Detailed information for materials and methods is provided in the Supplementary Data.

2.3. Total organic carbon and stable nitrogen/carbon isotope analyses

To directly compare the concentrations of SCCPs in contaminated soils from different studies, total organic carbon (TOC) in the individual soil samples was measured to normalize the concentrations of SCCPs. TOC in soil samples was measured using an O.I. Analytical Solids TOC Analyser (O.I. Analytical, USA). Before the TOC was measured, samples of approximately 0.1 g were treated with phosphoric acid ($1.0 \text{ mol} \cdot \text{L}^{-1}$) to remove inorganic carbon.

Stable nitrogen and carbon isotope analyses ($^{15}\text{N}/^{14}\text{N}_{\text{sample}}$ and $^{13}\text{C}/^{12}\text{C}_{\text{sample}}$) were performed on 1–2 mg of the homogenized biota samples using a Thermo DELTA V ADVANTAGE isotope ratio mass spectrometer interfaced with a Flash EA1112 HT elemental analyser (Thermo Fisher, Waltham, MA). Atmospheric nitrogen and PDB (Pee Dee Belemnite) were used as isotope ratio standards for N and C, respectively. Stable isotope abundances were expressed in parts per thousand (‰) according to the following equations:

$$\delta^{15}\text{N} = \left[\left(^{15}\text{N}/^{14}\text{N}_{\text{sample}} / ^{15}\text{N}/^{14}\text{N}_{\text{standard}} \right) - 1 \right] \times 1000 \quad (1)$$

$$\delta^{13}\text{C} = \left[\left(^{13}\text{C}/^{12}\text{C}_{\text{sample}} / ^{13}\text{C}/^{12}\text{C}_{\text{standard}} \right) - 1 \right] \times 1000. \quad (2)$$

The analytical precision for $\delta^{15}\text{N}$ was $\pm 0.2\text{‰}$, and the analytical precision for $\delta^{13}\text{C}$ was $\pm 0.1\text{‰}$.

2.4. Trophic levels (TLs) and biomagnification factors (BMFs)

Trophic levels (TLs) in biota samples were determined using the following equation (Hodum and Hobson, 2000):

$$\text{TL} = 2.3 + \left(\delta^{15}\text{N}_{\text{consumer}} - \delta^{15}\text{N}_{\text{Antarctic krill}} \right) / 3.3. \quad (3)$$

Biomagnification factors (BMFs) of the SCCPs between Agas and Ngas were calculated according to the following equation (Fisk et al., 2001):

$$\text{BMF} = [\text{predator}] / [\text{prey}] / (\text{TL predator} / \text{TL prey}) \quad (4)$$

where [predator] and [prey] are the lipid-normalized concentrations of SCCPs in Ngas and Agas samples, respectively.

2.5. Statistical analyses

Statistical analyses were performed using IBM SPSS Statistics Version 20 for Windows. The level of statistical significance was defined at $p < 0.05$. Mean concentrations and standard deviation for each species are listed in Table 1, after testing the normality of the data by One-Sample Kolmogorov-Smirnov Test. Difference in concentrations among species and locations was detected using one-way ANOVA with Student's *t*-test. Correlation analysis was conducted based on the calculation of Pearson's correlation coefficients between the tested variables. The regression analysis was only used for the variables found to be significant correlated to explore their quantitative relationship. Specifically, Pearson's correlation coefficients were calculated between SCCPs concentrations in organisms and lipid contents of the species as well as lipid-normalized concentrations of SCCPs and trophic levels. Linear regression analysis was then performed to explore the influence of the lipid contents on the SCCPs concentrations. The results for the statistical analyses were described in the Supplementary Data.

3. Results and discussion

3.1. SCCP concentrations

SCCPs were detected in all samples (Table S2), and detailed information for each species is listed in Table 1. To our knowledge, this is the first report of SCCPs concentrations in Antarctic biota samples. The occurrence of SCCPs in the different Antarctic matrices indicated that this type of organic pollutant was sufficiently persistent to travel from industrialized regions to polar regions. In the animal samples, the concentrations of SCCPs ranged from 40.2 to 256.6 ng/g dry weight (dw). In the vegetation samples, the concentrations of SCCPs ranged from 24.9 to 97.7 ng/g dw. For the contamination of soils, the range of SCCP concentrations was 3.5–32.8 ng/g dw. The concentrations of SCCPs in the animal samples were higher than those in the vegetation samples, and the lowest concentrations were found in the soil samples. Thus,

Table 1
Detailed parameters and concentrations of SCCPs in multiple matrices from the Antarctic.

Species	<i>n</i>	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Lipid content (%)	TOC (%)	ΣSCCPs (ng/g dw)	ΣSCCPs (ng/g ww)	ΣSCCPs (μg/g lw)
Animal samples								
Archeogastropoda (<i>Nacella concinna</i>)	16	-14.9 ± 1.3^a	5.6 ± 0.5	8.5 ± 1.3	N.T. ^b	123.6 ± 73.3	24.6 ± 14.6	1.5 ± 1.0
Neogastropoda (<i>Trophon geversianus</i>)	5	-16.0 ± 0.8	7.9 ± 0.3	4.8 ± 3.3	N.T.	88.9 ± 31.7	20.4 ± 7.3	2.7 ± 2.1
Fish (<i>Notothenia coriiceps</i>)	3	-18.1 ± 1.6	11.5 ± 0.2	5.7 ± 3.6	N.T.	69.9 ± 13.5	16.8 ± 3.2	1.5 ± 0.5
Vegetation samples								
Algae (<i>Halymenia floresia</i>)	6	-19.6 ± 2.9	-2.4 ± 9.3	3.4 ± 2.1	N.T.	61.3 ± 18.3	9.6 ± 2.9	2.8 ± 2.0
Moss (<i>Sanionia uncinata</i>)	6	-24.3 ± 1.6	1.8 ± 4.0	2.7 ± 0.9	N.T.	42.3 ± 15.2	8.5 ± 3.7	1.7 ± 0.7
Abiotic samples								
Soil	8	N.T.	N.T.	N.T.	8.8 ± 12.2	14.8 ± 11.4	N.T.	0.7 ± 0.6^c

^a Mean $\pm$ standard deviation (SD).

^b Normalized to total organic carbon (TOC).

^c Not tested.

the concentrations of SCCPs were higher in living organisms than those in the environmental substrate.

Based on comparisons of the SCCP concentrations with those of other studies, the concentrations of SCCPs were relatively low in the Antarctic animal samples, which were approximately one-tenth of those in mollusks from the Chinese Bohai Sea (109.6 ng/g vs. 1410 ng/g dw, respectively) (Yuan et al., 2012). The SCCP concentrations in the *Agas* (24.6 ng/g wet weight, ww), *Ngas* (20.4 ng/g ww), and fish samples (16.8 ng/g ww) were lower than those in marine mammal blubber from the Canadian Arctic (210 ng/g ww) (Tomy et al., 2000); however, the range of SCCP concentrations in the Antarctic biota was similar to that in biota from the European Arctic (5–88 ng/g ww) (Reth et al., 2006). Compared with the concentrations in Antarctic vegetation samples, Wang et al. (2015) found a much higher concentration of SCCPs in vegetation from Beijing, which varies from 980 to 1090 ng/g dw in bark and needle samples. The SCCP concentrations in Antarctic soil samples were lower than those detected in soils from the Liaohe River basin, China (14.8 vs. 114.7 ng/g dw, respectively) (Gao et al., 2012).

The concentrations of SCCPs in this study were much higher than those of other POPs in the Antarctic area. Borghini et al. (2005) reported that the concentrations of polychlorinated biphenyls (PCBs) were 0.4–0.6 ng/g dw in soil and 23–34 ng/g dw in moss from Victoria Land, Antarctica, which are values nearly twofold lower than the SCCP concentrations in our study. Wang et al. (2012) investigated the PCBs and the polybrominated diphenyl ethers (PBDEs) from an identical sample area, the mean concentration of PCBs was 0.4 ng/g dw in soil and 0.7 ng/g dw in moss, and the mean concentration of PBDEs was 0.024 ng/g dw in soil and 0.016 ng/g dw in moss. Compared with the SCCP concentrations in this study (14.8 ng/g dw in soil and 42.3 ng/g dw in moss), those concentrations are much lower. These comparisons all indicate that the SCCPs contamination is a primary component of the POP burden in Antarctica. Moreover, in regions other than Antarctica, SCCPs are also a dominant component of POP contamination; for example, in our previous studies, the concentration of SCCPs (Yuan et al., 2012) was higher than that of other POPs in samples from the Chinese Bohai Sea (Wang et al., 2008; Zhu et al., 2012). Thus, the global pollution of SCCPs requires further attention.

Fig. 2. Patterns of the mean percentages of the SCCP congener groups in six different matrices from Antarctica.

3.2. SCCP congener profiles

The SCCP congeners with 10 carbon atoms (C_{10} congeners) were dominant in all species, with proportions that varied between 49.1% and 69.0% (mean = 58.7%) of the total SCCP contamination and followed by C_{11} congeners (22.1%, mean), C_{12} congeners (11.1%, mean), and C_{13} congeners (8.2%, mean) (Fig. 2). Based on the number of chlorine, the SCCPs with 6 and 7 chlorine numbers (Cl_6 and Cl_7 congeners, accordingly) were the predominant congeners and contributed 39.6% and 32.7% to the total SCCP contamination, respectively. The SCCP congener pattern in our study was similar to that found in Arctic biota samples for which the congeners with shorter carbon chains (C_{10} congeners) and a lower degree of chlorination are also predominant (Tomy et al., 2000). However, the pattern of the SCCP congeners in this study was different from those in urbanized and industrial regions, which are dominated by relatively longer carbon chain congeners, such as C_{11} and C_{12} congeners (Ma et al., 2014b; Yuan et al., 2012; Zeng et al., 2011b).

The congener patterns were analysed in the aquatic and terrestrial biota. The Student's *t*-test confirmed that the proportion of C_{10} congeners was significantly higher in the aquatic biota (53.8% for Agas (except for site 3), 68.1% for Ngas, 69.0% for fish and 57.6% for algae) than that in the terrestrial biota (49.1% for moss) ($p = 0.028$, Supplementary Data-Results for statistical analysis R2). This can be explained by the fact that the C_{10} congeners are more hydrophilic than the longer chain congeners (Drouillard et al., 1998b; Feo et al., 2009), which make them easier to concentrate in aquatic organisms.

The proportion of C_{10} congeners was much higher in the Antarctic samples than that reported in previous studies (Fig. 3) (Borgen et al., 2002; Houde et al., 2008; Ma et al., 2014a; Reth et al., 2006; Tomy et al., 2000; Zeng et al., 2011b). The high volatility and water solubility of shorter carbon chain congeners might explain the higher proportion of C_{10} congeners in Antarctic samples. The volatility of SCCPs decreases with an increase in the carbon chain length (Drouillard et al., 1998a). C_{10} congeners have high volatility, but they also have low affinity for atmospheric particulates, which combined result in a strong potential for long-range atmospheric transport. Ma et al. (2014c) also found that C_{10} congeners predominated in Antarctic gas-phase air samples. Additionally, oceanic transport influences the transfer and deposition of pollutants at a global scale (Farrington and Takada, 2014; Dachs et al., 2002). Therefore, the C_{10} congeners of SCCPs, with lower *K*_{ow} and higher water solubility values than other congeners, are more difficult for sediments to absorb and can migrate easily with ocean currents for long distances (Feo et al., 2009). The different patterns of congener

groups also indicated that long-range transport might be the primary source of the SCCP burden in Antarctica.

3.3. Spatial distribution of SCCPs

Local human activities that include scientific expedition, tourism, and airport service might increase anthropogenic pollution in Antarctica. As shown in Fig. 1, Agas collected at sites 3, 4, and 16 had relatively higher concentrations of SCCPs than those of other sites, with a mean level of 226.5 ng/g dw. By contrast, the mean concentration of SCCPs was 83.6 ng/g dw in Agas collected on the southern part of the Fildes Peninsula (specifically, sites 12, 13, and 14), which is far from human activities. The Student's *t*-test found that the concentrations of SCCPs in the Agas samples collected near the research station were significantly higher than those in the Agas samples from the remote sample sites ($p = 3 \times 10^{-6}$, Supplementary Data-R3). Additionally, the patterns of the SCCP congener groups were also different in the Agas collected from the different areas (Fig. S2). In the Agas from the remote sample sites, most of the SCCPs were C_{10} congeners (mean = 60.6%), but the pattern of congener groups near the research station (site 3) was similar to that for an industrial area (Ma et al., 2014a), with percentages of the C_{10} – C_{13} groups of 28.2%, 36.6%, 22.0% and 13.3%, respectively. The different levels of concentration and patterns of congener groups between those of the research station areas and those of the remote areas indicated that anthropogenic activities were a significant factor in the spatial distribution of SCCPs on the Fildes Peninsula.

The population density is very low on the Fildes Peninsula, and the stations in Antarctica all have efficient waste treatment systems to minimize their influence on the local ecosystems. Wild et al. (2015) studied the influence of scientific stations in Antarctica and found that the levels of PBDEs and perfluoroalkylated substances (PFASs) rapidly decreased within 200–2000 m of the station areas. Our data also showed a sharp decrease (63.1%: from station areas to remote areas) in the concentrations of SCCPs in Agas samples, and therefore, the contamination that originated locally in Antarctica likely had little influence on the areas far from human activities. Besides, the congener patterns of SCCPs in the Antarctic samples collected from the remote sites showed a unique feature compared with those of mid- and low-latitude samples and technical mixtures (Fig. 3): the proportion of C_{10} congeners was higher. It is not likely that pollution of local origin explains this different SCCP congener pattern. Based on the results, long-range transport is more likely to be the primary source of SCCPs in the Antarctic organisms, particularly for those collected far from the centres of human activity.

3.4. Bioaccumulation and biomagnification of SCCPs

Significant correlation between the concentrations of SCCPs and the contents of lipids in biota samples was detected by Pearson correlation analysis ($p = 0.014$, $N = 36$). Further regression analysis revealed that the concentrations of SCCPs in biota was linearly dependent on the lipid content (SCCPs concentrations ng/g dw = $(7.8 \pm 3.0) \times \text{Lipid contents} + (43.8 \pm 20.2)$, $r^2 = 0.14$, $p = 0.014$, $N = 36$), indicating the great influence of the lipid contents on the SCCPs bioaccumulation because of the hydrophobicity of the pollutants (Fig. S3 and Supplementary Data-R4). Additionally, it was notable that the marine animal species with a higher lipid content had a higher proportion of longer chain congeners than that of the lower lipid-content species (Table S3A and Supplementary Data-R5). According to Feo et al. (2009), the log *K*_{ow} of SCCP congeners increases with an increase in carbon chain length, which leads to the preferential enrichment of longer carbon chain congeners (higher *K*_{ow} values) in lipids. In this study, enrichment of the more hydrophobic SCCP congeners also occurred in the species with higher lipid contents (Table S3B).

In aquatic food webs, many POPs can biomagnify or show an increased lipid-normalized concentration from low trophic level species to higher trophic level species (Fisk et al., 2001). In the present study,

Fig. 3. Comparison of the C_{10} , C_{11} , C_{12} and C_{13} congener group patterns in matrices from Antarctica with those of other studies from different sample areas and technical mixtures (a: Technical mixtures and biota including Arctic char (*Salvelinus alpinus*), cod (*Gadus morhua*), little auk (*Rissa tridactyla*) and kittiwake (*Alle alle*), Reth et al., 2006; b: Borgen et al., 2002; c: Aquatic species including leather catfish (*Silurus meridionalis*), common carp (*Cyprinus carpio*), Chinese softshell turtles (*Chinemys reevesii*) and java tilapia (*Tilapia nilotica*), Zeng et al., 2011b; d: Houde et al., 2008; e: Ma et al., 2014a; f: Tomy et al., 2000).

no significant correlation was found between the TL values and the lipid-normalized concentrations of SCCPs in the aquatic samples ($p = 0.57$, $N = 28$; Supplementary Data-R6). However, in previous studies, significant positive correlations between the TL values and the lipid-normalized concentrations of SCCPs were found in the aquatic food chains (Ma et al., 2014b; Zeng et al., 2011b). When considering the influence of a direct predator-prey relationship on the biomagnification of SCCPs, a positive correlation was expected between the predator-prey species in Antarctica. Fortunately, field photos taken during the sampling for this study provided support for the direct predator-prey relationship between the *Agas* and *Ngas* species (field photos in Fig. S1: the *Ngas* is eating the *Agas*). Moreover, earlier lab-reared studies reported that consumers had stable carbon isotope ratios ($\delta^{13}\text{C}$) that were not significantly different from those of their food ($\Delta\delta^{13}\text{C} = 0.8 \pm 1.1\%$ SD), whereas the consumers were enriched in ^{15}N relative to their diet ($\Delta\delta^{15}\text{N} = 3.0 \pm 2.6\%$ SD) (Jardine et al., 2006). In the current study, the values of $\Delta\delta^{13}\text{C}$ and $\Delta\delta^{15}\text{N}$ (mean differences for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$, respectively) between *Agas* and *Ngas* samples were 1.1‰ and 2.3‰, respectively, which were in the range of the corresponding criteria ($\Delta\delta^{13}\text{C} = 0.8 \pm 1.1\%$ SD; $\Delta\delta^{15}\text{N} = 3.0 \pm 2.6\%$ SD, Jardine et al., 2006). These results confirmed the predator-prey relationship between the two species, although other food sources may be consumed (Harper and Peck, 2003). No significant correlation between the TL values and lipid-normalized concentrations of SCCPs in *Agas* and *Ngas* samples was detected ($p = 0.17$, $N = 21$, Supplementary Data-R7), which indicated that the complex interplay of physical-chemical and biological processes effect the trophic transfer potential of SCCPs in the Antarctic organisms.

The calculated biomagnification factors (BMFs) of the SCCPs between *Ngas* and *Agas* ranged from 0.4 to 3.5, with a mean of 1.9 (Table S4), which is a range consistent with that in the food webs from Lake Ontario and Lake Michigan (0.5–3.6) and indicates that SCCPs could biomagnify between prey and predators (Houde et al., 2008). However, these values were relatively low compared with the trophic magnification factors (TMFs) of other POPs, such as for perfluorooctane sulfonate in a food web from Lake Ontario (TMF = 5.9) (Martin et al., 2004) and for hexabromocyclododecane in a food web from Lake Ontario (TMF = 6.3) (Tomy et al., 2004). The biotransformation half-life of SCCPs (9.8–93.9 d, Table S5) was shorter than that for PCBs (11.2–1647 d), as calculated by the EPI Suite (US EPA, 2012), and indicated that SCCPs are more easily biodegraded than PCBs. Thus, these results indicate that the biomagnification potential of SCCPs through a food chain might be lower than that of other POPs.

Fig. 4. BMFs for the different SCCP congeners in Antarctic *Agas* and *Ngas* species collected from the identical locations.

The BMFs for the individual congener groups (C_{10} – C_{13}) ranged from 0.1 to 4.0 between the *Agas* and *Ngas* species (Fig. 4). The average BMF of the SCCPs decreased from C_{10} to C_{13} congeners, with values of 2.3, 1.8, 1.3, and 0.8, respectively. A similar pattern of decreasing BMFs with the increasing carbon chain length of SCCP congeners was observed in a food web from Lake Michigan (Houde et al., 2008). However, in previous laboratory exposure experiments, the BMFs tended to increase with increasing lengths of the carbon chain (Fisk et al., 1998; Fisk et al., 2000). Notably, the biomagnification of shorter chain congeners of SCCPs was higher than that of the longer chain ones in the field study, which might be attributed to the larger molecular sizes of the longer carbon chain congeners, as Mackintosh et al. (2004) found that the concentrations of high molecular-weight phthalate esters declined significantly at higher trophic levels in a marine food web from western Canada.

Lavoie et al. (2013) reported that the biomagnification factors of mercury are the highest ones in cold and low productivity systems through food webs. The average temperature was extremely low and turned out to be approximately $-3\text{ }^{\circ}\text{C}$ on the Fildes Peninsula at King George Island, Antarctica. Therefore, large BMF values are expected for the SCCPs in the Antarctic biota. Unfortunately, there is no BMF information for SCCPs reported elsewhere for the same predator-prey species in Antarctica, *Ngas* and *Agas*. Limited information showed the BMF of SCCPs was 0.9 and 1.2 for the invertebrate–fish food webs in Lake Ontario and Lake Michigan with the temperatures of 16 – $21\text{ }^{\circ}\text{C}$ (Houde et al., 2008). A BMF value of 1.6 was observed for different fish species in a cooling-water recipient lake in China with the temperatures above $30\text{ }^{\circ}\text{C}$ (Zeng et al. 2011b). The BMF of SCCPs was 2.4 for the zooplankton–shrimp–fish food web in Liaodong Bay with an annual average temperature of approximately $10\text{ }^{\circ}\text{C}$ (Ma et al., 2014b). Clearly, it is impossible to find a clue as to whether or not the biomagnification potential of SCCPs show a temperature-dependent manner due to the species difference. The temperature effect on the biomagnification potential of SCCPs needs further investigation.

4. Conclusions

The concentrations of SCCPs in the Antarctic samples of this study were lower than those of the mid- and low-latitude regions. Although human activities affected the local distribution of SCCPs, the long-range transport behaviour of SCCPs was confirmed by the detection of SCCPs in Antarctic remote areas. Additionally, the enrichment of C_{10} congeners indicated a high potential for the long-range transport of shorter chain congeners. SCCPs were enriched in the species with high lipid contents, and the potential for biomagnification was found between *Agas* and *Ngas* species. Most organisms in polar regions are endemics and may be sensitive and vulnerable to the adverse effects of persistent contaminants; therefore, more studies should focus on the potential bioaccumulation and toxicological risks of POPs in polar environments.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <http://dx.doi.org/10.1016/j.envint.2016.05.005>.

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