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EDITOR'S NOTE:

This paper is 1 of 3 articles resulting from a workshop sponsored by The Health and Environmental Sciences Institute (HESI) held in January 2013 in Miami, Florida, USA. The aim of the workshop was to review current practices, identify data gaps, and provide recommendations to improve current methods and develop new methods supporting both prospective and retrospective environmental assessments of organic chemical bioaccumulation in terrestrial ecosystems.

ABSTRACT

Field-based studies are an essential component of research addressing the behavior of organic chemicals, and a unique line of evidence that can be used to assess bioaccumulation potential in chemical registration programs and aid in development of associated laboratory and modeling efforts. To aid scientific and regulatory discourse on the application of terrestrial field data in this manner, this article provides practical recommendations regarding the generation and interpretation of terrestrial field data. Currently, biota-to-soil-accumulation factors (BSAFs), biomagnification factors (BMFs), and bioaccumulation factors (BAFs) are the most suitable bioaccumulation metrics that are applicable to bioaccumulation assessment evaluations and able to be generated from terrestrial field studies with relatively low uncertainty. Biomagnification factors calculated from field-collected samples of terrestrial carnivores and their prey appear to be particularly robust indicators of bioaccumulation potential. The use of stable isotope ratios for quantification of trophic relationships in terrestrial ecosystems needs to be further developed to resolve uncertainties associated with the calculation of terrestrial trophic magnification factors (TMFs). Sampling efforts for terrestrial field studies should strive for efficiency, and advice on optimization of study sample sizes, practical considerations for obtaining samples, selection of tissues for analysis, and data interpretation is provided. Although there is still much to be learned regarding terrestrial bioaccumulation, these recommendations provide some initial guidance to the present application of terrestrial field data as a line of evidence in the assessment of chemical bioaccumulation potential and a resource to inform laboratory and modeling efforts. *Integr Environ Assess Manag* 2016;12:135–145. © 2015 The Authors. *Integrated Environmental Assessment and Management* published by Wiley Periodicals, Inc. on behalf of SETAC.

Keywords: Biomagnification factors Biota-to-soil-accumulation factors BMF BSAF Chemical bioaccumulation Terrestrial food web TMF Trophic magnification factors

INTRODUCTION

The potential of compounds to bioaccumulate in organisms and to transfer and biomagnify in food webs is a key consideration in chemical regulation (Weisbrod et al. 2009). Currently, the assessment of the bioaccumulation potential is primarily based on data derived from marine or freshwater organisms and food webs, and many assessments include field data collected from wild aquatic organisms. However, physiological and ecological factors affecting bioaccumulation in terrestrial ecosystems are considered to be very different from those in aquatic ecosystems. Hence, bioaccumulation

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assessments derived from aquatic systems may not be predictive of bioaccumulation potential in terrestrial systems (Kelly and Gobas 2001, 2003; Kelly et al. 2007).

It is generally understood that soil properties such as organic C content and quality affect the bioavailability of chemicals, and thus uptake of organic compounds by soil organisms (Chung and Alexander 2002; Amorim et al. 2005; Cornelissen et al. 2005). Furthermore, the availability of organic compounds in soil may decrease over time due to aging and consecutive increased binding of the chemical to soil particles and weathering or degradation of the compound (Belfroid et al. 1995; Styrishave et al. 2008; Johnson, Salice et al. 2009). Current methods used to predict bioaccumulation potential of organics in aquatic systems rely on measures of hydrophobicity and coefficients such as $\log K_{OW}$. However, in terrestrial systems, $\log K_{OW}$ alone does not explain or predict bioaccumulation (Belfroid et al. 1995). In terrestrial systems, biotransformation seems to have more profound effects on bioaccumulation and biomagnification in food webs (Kelly et al. 2007; McLachlan et al. 2011) (see Supplemental Data for more details).

Despite the possible discrepancy between bioaccumulation in aquatic and terrestrial ecosystems, explicit assessment of terrestrial bioaccumulation data are not specified in national legislations or specifically required in bioaccumulation assessments. In the European Union (EU), the amendment of Annex XIII in the current Regulation on Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) requires consideration of all available bioaccumulation metrics as part of a weight of evidence analysis. As a result, terrestrial field studies or comparable laboratory simulations, if available, are recommended for EU chemical assessments (Moermond et al. 2012; Vierke et al. 2012; Gottardo et al. 2014), although formal guidance or recommendations on the use of terrestrial bioaccumulation data and thresholds are not available.

In January 2013, the ILSI Health and Environmental Sciences Institute (HESI) sponsored a workshop on terrestrial bioaccumulation in Miami, Florida, USA. The goal of the workshop was to compile information and inform upon a framework for the assessment of bioaccumulation in terrestrial food webs that would be useful in chemical registration programs. This article provides an overview on the different approaches to evaluate organic chemicals data generated from wild terrestrial organisms and abiotic media collected from field investigations. This article also addresses the advantages and opportunities for using terrestrial field data in bioaccumulation assessments and provides practical recommendations for generating and applying such data. Companion articles provide similar focus on the use of data generated from terrestrial laboratory studies (Hoke et al. this issue) and environmental modeling (Gobas et al. this issue).

Importance terrestrial field data in bioaccumulation assessments

Measurement of organic chemicals in aquatic organisms and abiotic media provide important data that have been used to evaluate bioaccumulation potential (Selck et al. 2012). Depending on the types of information collected, chemical concentration data from field-collected samples can be used to calculate bioaccumulation factors (BAFs), biomagnification factors (BMFs), and trophic magnification factors (TMFs) (Gobas et al. 2009; Conder et al. 2012). Bioaccumulation data from terrestrial settings can also be easily expressed in this manner, including biota-soil accumulation factors (BSAFs), BAFs, BMFs, and TMFs (Gobas et al. this issue). When derived in terrestrial field studies, these metrics can be used in a weight of evidence approach to assess the bioaccumulation potential of chemicals that have been released in the environment in similar approaches as those used to evaluate data from aquatic bioaccumulation potential assessments. In addition, such data can be used to inform the development, validation, and refinement of laboratory tests and models for prospective assessments of chemicals that have yet to be released into the environment.

There is a wealth of terrestrial field data that can be used to assess bioaccumulation potential (see Supplemental Data for >20 different studies). The available evidence strongly suggests that terrestrial field data provide information that is not always consistent with data generated from aquatic studies. For example, field data compiled for 4 example chemicals, PCB-153, pyrene, and perfluorooctane sulfonic acid (PFOS), demonstrate both the utility of and need for consideration of terrestrial data in the regulatory assessment of bioaccumulation potential (Figure 1; for details on the studies and the derivation of the metrics, see the Supplemental Data). In the case of PCB-153, the BCFs derived from aquatic studies imply high potential for accumulation, but the BSAFs for invertebrates and plants (approximate terrestrial analogues to aquatic BCFs) indicate a much lower potential to bioaccumulate in terrestrial systems (Figure 1A). Furthermore, the examples indicate that aquatic derived BCFs and BAFs do not always match estimates of bioaccumulation potential derived from terrestrial (soil) organisms or avian and mammalian species. For pyrene, a metabolizable polycyclic aromatic hydrocarbon, bioaccumulation potential is supported by aquatic BCF and BAF estimates, but not by 1) BSAFs for terrestrial invertebrates and plants (generally <1) or 2) BMF-data from homeothermic animals in terrestrial and aquatic food webs (Figure 1B). The converse (bioaccumulation potential not indicated by aquatic BCFs and BAFs but instead by BMF data for homeotherms) is found with PFOS (Figure 1C).

Recommendations for generating field study data useful to terrestrial bioaccumulation assessments

Given that terrestrial field data have value in providing useful evidence in bioaccumulation potential assessment for particular chemicals, as well as information useful to modeling and laboratory methods in support of bioaccumulation assessments, workshop participants were able to provide some initial practical recommendations regarding the interpretation and generation of terrestrial field data. Workshop participants focused on several primary issues, as detailed in the remainder of this section: the selection of species and tissue types to include in investigations or to focus upon in existing data sets; considerations for spatial and temporal aspects of sampling and data analysis; available methods to determine food web relationships; considerations for sample sizes needed for robust bioaccumulation data analysis; and general practical advice on obtaining samples from terrestrial organisms. The guidance provided may need to be refined as advances are made in regulatory and technical aspects related to the assessment of bioaccumulation potential.

Selection of species. Central to species selection is the need to identify trophic guilds and predator–prey interactions that can generate data reflecting key food web relationships.

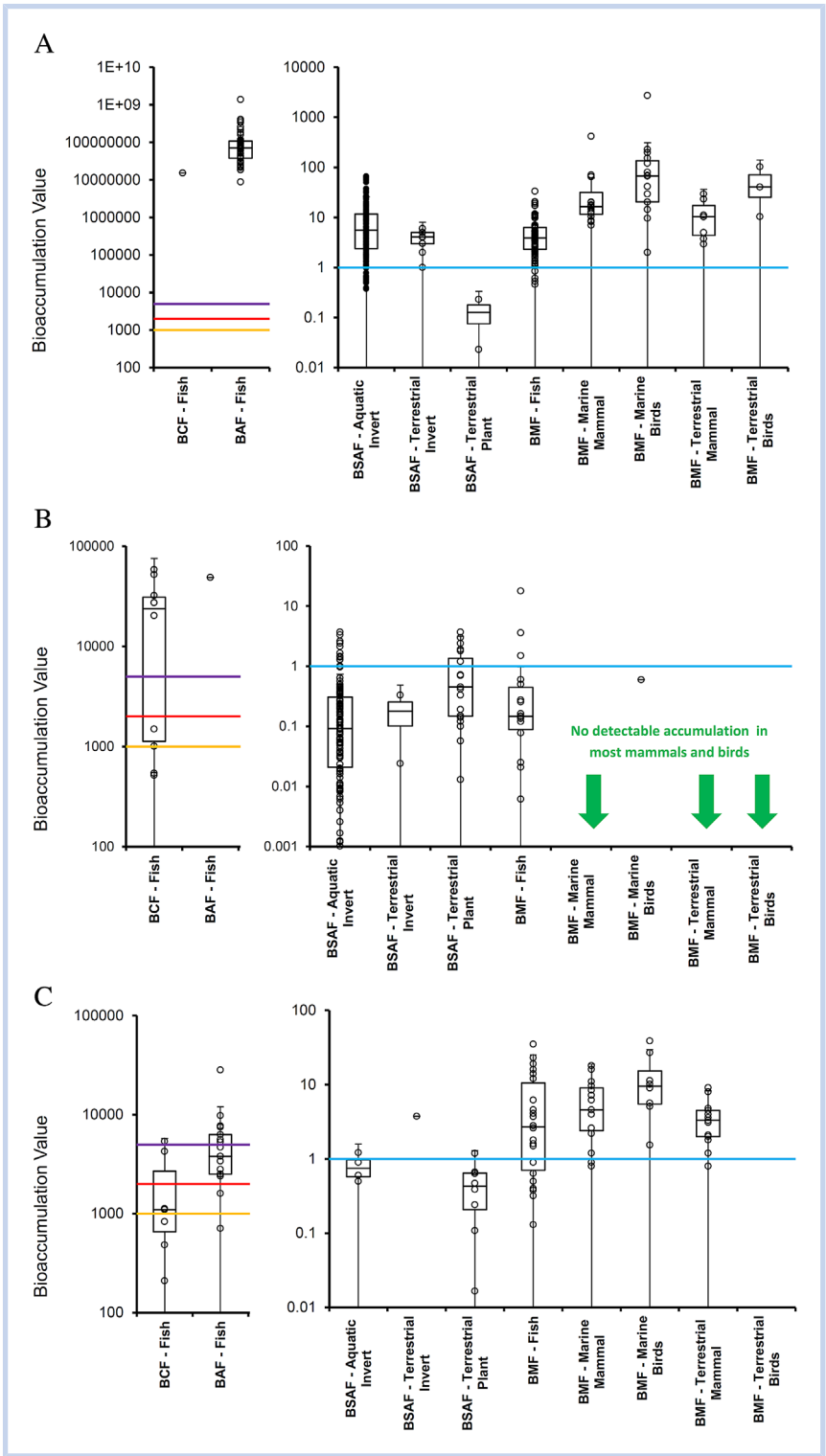

Figure 1. Bioaccumulation metric values for PCB-153 (**A**), pyrene (**B**), and PFOS (**C**). Line markers include one, the scientific definition of bioaccumulation (blue line); and values of 1000 (orange line), 2000 (red line), and 5000 (purple line) associated with various bioaccumulation levels of concern for US, Canadian, and/or European regulatory agencies.

Measurements of plant tissues and soft-bodied soil invertebrates should be used to generate BSAFs, because these organisms are in direct contact with the soil. The use of BSAFs is not recommended for estimating bioaccumulation potential in higher trophic level vertebrate species (i.e., reptiles, birds, and mammals) because of the considerable uncertainties associated with exposure and uptake. The primary route of organic chemical exposure for these organisms is generally dietary (USEPA 1997, 2005), with only a minor contribution from soil ingestion (Beyer et al. 1994). Consequently, chemical bioaccumulation for these species in terrestrial systems is better represented by BMFs or TMFs.

Calculation of BMF values requires concentrations of chemical in a predator (or consumer) and its diet, usually represented by one or more food items. Selection of a predator–prey pair to measure is an option that should be considered in

experimental design and interpretation of BMFs from trophic level data sets. It is clear that BMF values in aquatic systems vary among trophic levels (Conder et al. 2012), suggesting that a particular species or component of a terrestrial food web may also be a more robust indicator of bioaccumulation. To evaluate this hypothesis, BMFs and BSAFs for hydrophobic chemicals typically considered bioaccumulative (e.g., PCBs, PBDEs, and DDTs) were compiled from 8 studies that evaluated chemical residues in terrestrial invertebrates (earthworms) exposed in the laboratory to field-contaminated soils, and matched predator–diet pairs for mammalian insectivore, omnivore, and carnivore species as well as avian carnivore species (Hebert et al. 1994; Belfroid et al. 1995; Harris et al. 2000; Kelly and Gobas 2001; Matscheko et al. 2002; Blankenship et al. 2005; Voorspoels et al. 2007; White et al. 2007) (see Supplemental Data). These data suggest that soil invertebrates and avian carnivores provide the most compelling measures of bioaccumulation potential (lowest variance coefficient, Figure 2).

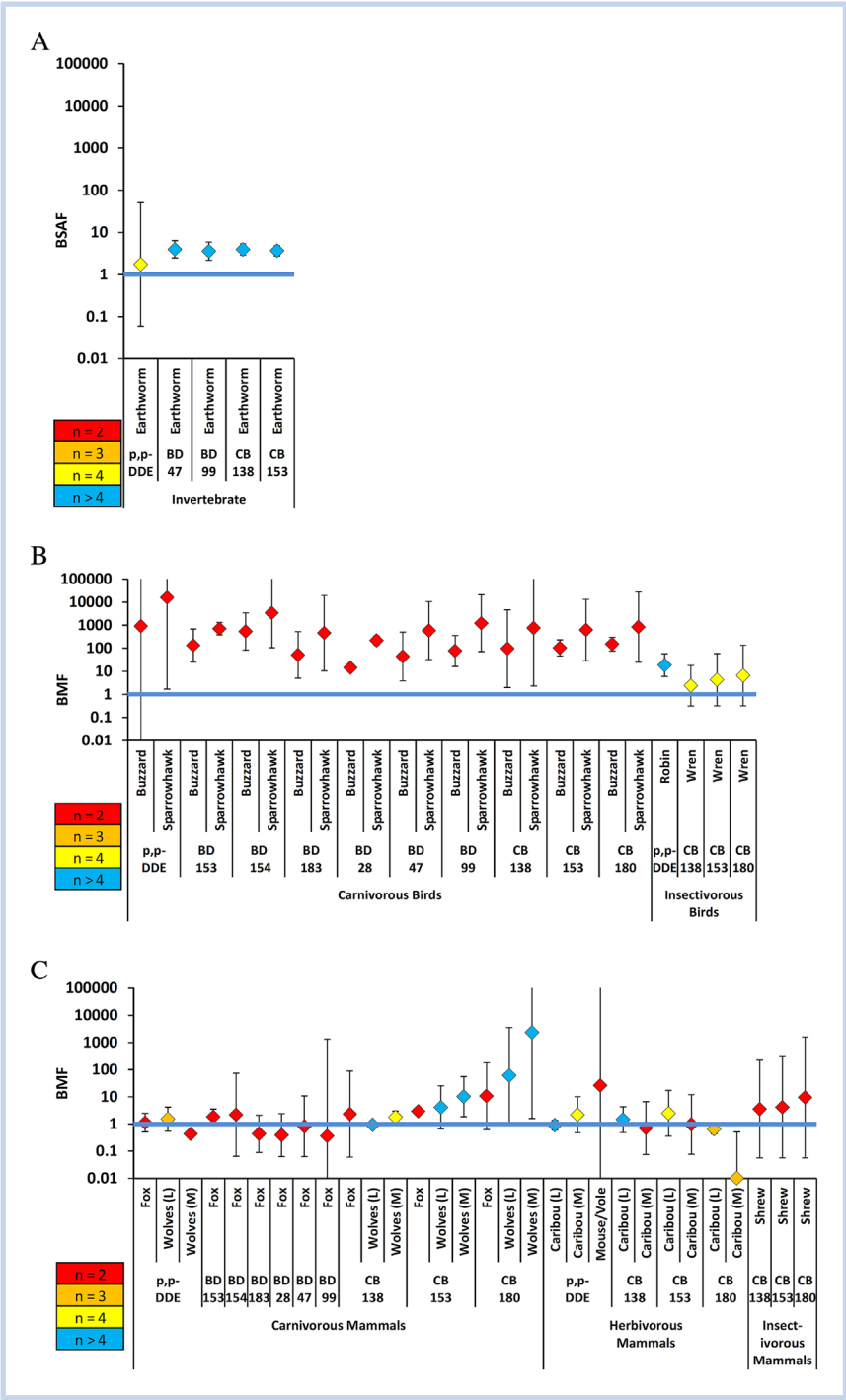

Figure 2. Mean (95% CI) BSAFs (g, soil OC/g, lipid) for soil invertebrates (**A**), BMFs (g, diet lipid/g, predator lipid) for birds (**B**) and BMFs for mammals (**C**) associated with p,p-DDE, PCBs congeners (noted "CB" plus congener number), and PDBE congeners (noted "BD" plus congener number). Blue line indicates a value of 1 (threshold for bioaccumulation potential). "L" and "M" labels note BMFs developed using liver and muscle sample data, respectively. Other values were based on averages of BMFs based on liver and muscle tissues (e.g., buzzard, sparrowhawk, and fox), tissue samples not identified by the source study (e.g., shrew, wren, robin), or soil-voided whole body and soil measurements (e.g., earthworms).

Approximately 80% of invertebrate BSAFs (Figure 2A) and avian BMFs (Figure 2B) were greater than 1 for bioaccumulative chemicals, compared to less than 20% of the BMFs for mammalian species (Figure 2C). The data indicate that carnivores exhibited greater bioaccumulation potential than other guilds and, thus, carnivores may be the most appropriate sentinel species for measuring bioaccumulation potential in terrestrial field studies. For example, data for avian invertivores tended to indicate bioaccumulation less often (only 1 of 4 avian invertivore BMFs indicated bioaccumulation potential) than data for avian carnivores (Figure 2B), and only 5 of the 30 BMFs for carnivorous mammals were indicative of bioaccumulation (Figure 2C).

Substantially greater BMF values for carnivores are not unexpected, as noted in modeling and empirical studies (Kelly and Gobas 2003; Debruyn and Gobas 2006; Kelly et al. 2007). Carnivores tend to exhibit high BMFs due to their top (and sometimes apex) position in terrestrial food webs (Kelly and Gobas 2003; Kelly et al. 2007). BMFs may increase for predators situated at the top of food webs due to a great number of trophic transfers from lower tier predators and prey species (Debruyn and Gobas 2006). Greater bioaccumulation in avian species compared to mammalian predators has been observed in other field studies of aquatic food webs (Hop et al. 2002; Hallanger et al. 2011), suggesting that avian carnivores may be an important guild to include in terrestrial studies evaluating bioaccumulation potential of chemicals. Larger avian BMFs could be related to a number of physiological or ecological differences between birds and mammals. For example, fish-eating birds were shown to have less cytochrome P450-associated monooxygenase activity compared to mammals (Walker 1980), leading to less metabolism and greater accumulation of chemicals. This may be related to a relatively low exposure to complex plant secondary metabolites and a lack of evolutionary selection for the capacity to detoxify these compounds. Avian species also require less water intake compared to mammals (Sample et al. 1997), and metabolites in urine can be re-absorbed in the cloaca (Walker 1983), possibly resulting in reduced urinary excretion pathways for chemicals. For terrestrial species, the respiratory elimination route is also of importance with respect to chemical accumulation (Kelly et al. 2007). However, the avian respiratory pathway is very efficient in O_2–CO_2 exchange, resulting in lower breathing rates in comparison to mammals (normalized to body weight) (Sample et al. 1997). Such lower breathing rates may result in less respiratory elimination of organic contaminants. However, it is clear that more modeling and empirical work is needed to evaluate the hypothesis that avian carnivores may be top bioaccumulators in most terrestrial food webs.

Selection of tissue type. Selection of tissue types and treatment of samples should be carefully evaluated for terrestrial bioaccumulation determinations. Field data used to derive BSAFs for earthworms should ideally be based on depurated (soil-free) whole body analyses. Chemicals associated with soil within the gut of earthworms have not been transferred to the biological compartment and, thus, do not truly represent the process of bioaccumulation. However, if earthworms are used to estimate a BMF, as prey item, this depuration process may not be appropriate because predators consume nondepurated prey. Plant-based BSAFs are usually established on the analysis of aboveground tissues such as leaves or fruits, because those tissues are generally consumed by higher trophic level animals. However, BSAFs generated from aboveground tissues are generally lower than those based on roots due to preferential partitioning of nonionic organic chemicals to roots than other plant tissues (Simonich and Hites 1995; Collins et al. 2005). Plant and fruit BSAFs can be affected by aerial deposition of organics onto aboveground surfaces, and therefore, inaccurately represent the bioaccumulation process from soil.

In general, matched predator–diet samples used to calculate BMF values are more difficult to obtain compared to samples needed for BSAFs, and BMF data interpretation is more difficult. Several factors influence the collection and interpretation of BMF data, including spatial variation, behavior, habitat, time of year, reproductive status, and other characteristics of predator and diet (Borga et al. 2012). The tissue type targeted for sampling and testing also is an important consideration because it is rarely practical to analyze whole organisms, especially for larger predators. Theory suggests that concentrations of hydrophobic compounds in tissues can be lipid-normalized to account for differences in fugacity due to different lipid contents. If so, BMFs based on lipid normalized concentrations should be similar between tissue types and provide similar information for organic compounds. To demonstrate this important consideration, studies that included the concentrations of organic chemicals in both the liver and muscle tissues of organisms from terrestrial food webs were compiled to calculate paired BMF_{liver} and BMF_{muscle} values (see Supplemental Data). Figure 3A indicates that BMF_{muscle} and BMF_{liver} are related to each other and not significantly different for legacy, nonpolar organic contaminants. Nevertheless, even for nonpolar organic chemicals, rapid changes in the body condition of the organism may result in internal remobilization of chemicals, which will disturb the internal equilibrium (Crosse et al. 2013). In such cases, even lipid-normalized chemical concentrations may not be comparable between tissue types. Organic contaminants that do not preferentially partition into lipids (e.g., some perfluoroalkyl and polyfluoroalkyl substances [PFASs]), cannot be lipid normalized to account for differences in tissue concentration, and present challenges for evaluating bioaccumulation via individual tissues. For example, individual tissue BMFs vary widely for some PFASs due to differences in bioaccumulation patterns that are not yet understood (see Figure 3B, $r = -0.06$ and $p = 0.77$). To accommodate this uncertainty, BMFs are often evaluated on a whole body basis by estimating the concentration in the entire body on the basis of an organ mass balance and measurement of PFASs in several different tissues (Houde, Martin et al. 2006; Müller et al. 2011). Overall, BMF_{muscle} showed better agreement with $BMF_{wholebody}$ than did BMF_{liver} for both wolves and caribou (Figure S1). Although whole body BMF values are preferred metrics for assessing bioaccumulation of PFASs, BMF_{muscle} may appear to be an acceptable surrogate for this specific class of chemicals.

Selection of sampling location and timing. Both spatial and temporal variation of chemical contaminants should be taken into account in terrestrial ecosystems when addressing bioaccumulation of compounds under field conditions. The first source of spatial variation is the spatial distribution of substances in soil, which may be related to the primary source of the compounds, their dispersal, and both soil and chemical properties (Heywood et al. 2006). Within-site variation in soil concentrations of organic compounds can be substantial (up to

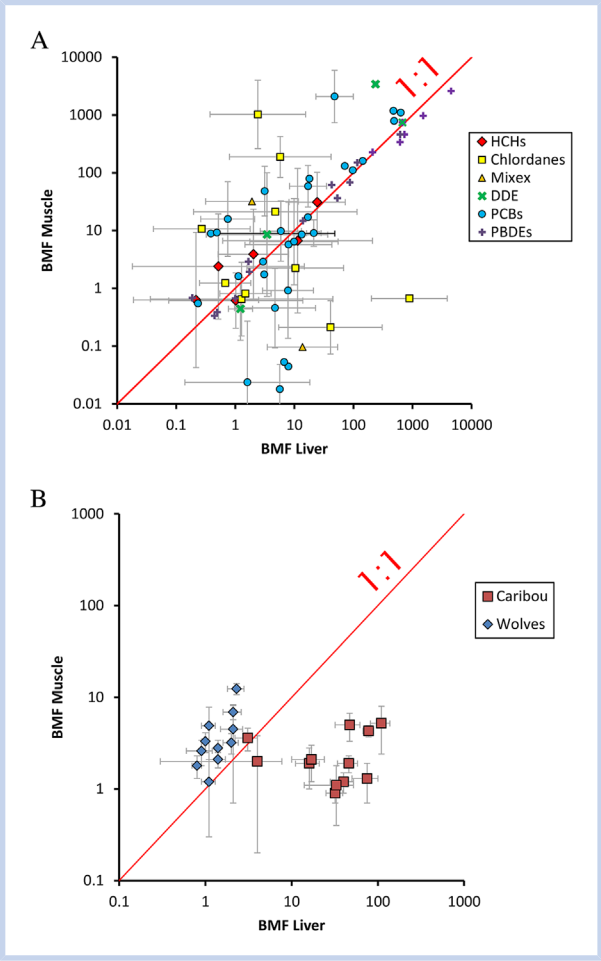

Figure 3. Correlation between BMF_{Liver} and BMF_{Muscle}. (**A**) Data on nonpolar organic contaminants from different species. (**B**) Data on PFASs from caribou and wolves. Error bars represent 1 SD of the mean. Lines indicate a 1:1 relationship. Data from Kelly and Gobas (2001), Müller et al. (2011), and Voorspoels et al. (2007).

3 orders of magnitude) (Johnson, Korcz et al. 2009; Niemeyer et al. 2010) and will greatly influence bioaccumulation. A second source of spatio-temporal variation is the availability of prey items. For example, prey availability for the little owl (*Athene noctua*) varied significantly during season and among habitat types such that modeled chemical exposures varied by 3 orders of magnitude (Schipper et al. 2012). In a field study with small mammals, species-specific spatial habitat usage and diet composition also greatly influenced chemical uptake, in addition to metal bioavailability (van den Brink et al. 2011). Such environmental and ecological factors may influence bioaccumulation under field conditions for some chemicals to such extend that BMFs for particular predator–prey relationships may differ greatly from BMFs determined in other seasons or dissimilar habitats.

Evidence of spatio–temporal variability from these and other studies suggest that when calculating a BMF, it may be preferable to evaluate the locations of prey capture relative to the predator's foraging range. If predators obtain most prey from a relatively small but highly contaminated area comprising a tiny fraction of its foraging range, this may bias the calculation of the BMF. It is more appropriate to target the collection of prey from an area defined by its predator's foraging range, rather than focusing on the footprint of the total contaminated area. Additionally, it may be advantageous to collect several different prey items, particularly if the predator diet varies seasonally, to derive a time-weighted average concentration in the diet.

Assessment of food web relationships. Determining BMF and TMF metrics that can be used to evaluate bioaccumulation potential requires information on the food web and trophic position to evaluate field data and design relevant and meaningful field studies. The 2 most widely used techniques to quantify these ecological factors are analysis of stable isotopes in tissues and food items and dietary characterization via gut or fecal analysis. A third technique showing promise is the use of molecular level analysis, although this approach has yet to be widely used in ecotoxicology studies.

At present, stable isotopes of N ($^{15}N/^{14}N$; $\delta^{15}N$) are widely used to characterize trophic position of terrestrial organisms, whereas C ($^{13}C/^{12}C$; $\delta^{13}C$) and S ($^{34}S/^{32}S$; $\delta^{34}S$) have been applied to characterize diets (Peterson and Fry 1987; Kelly 2000; Koch 2008). Sulphur has been less widely used because few laboratories routinely conduct this isotope ratio analysis. Increases in $\delta^{15}N$ occur because of the preferential retention of the heavier isotope from the diet of the consumer. This fractionation is related to excretion of urea and other nitrogenous substances that are enriched in ^{14}N relative to body N pools (Parker et al. 2005). In aquatic environments, this fractionation is relatively constant with an enrichment factor ($\delta^{15}N$) of 3.0‰ to 5.0‰ between trophic levels. The $\delta^{15}N$, used to calculate trophic level (TL), is often assumed to be 3.4‰ to 3.8‰ based on a number of feeding experiments or syntheses of the literature (Hobson and Welch 1992; Post 2002; Jardine et al. 2006). However, feeding experiments on birds and mammals have shown that the magnitude of fractionation increases with increasing protein content in the diet, possibly because animals on low protein diets use most of their dietary N for protein synthesis, and consequently have a lower urea N flux (Koch 2008). Metabolic differences between taxa may also be important. For example, the $\delta^{15}N$ between an avian diet and its muscle tissue was only 2.4‰ (Mizutani et al. 1991). This has implications for using a single $\delta^{15}N$ to estimate the trophic position of organisms within terrestrial avian and mammalian food webs and contributes uncertainty in the calculation of the trophic enrichment factor within food webs.

In addition, environmental factors such as precipitation, temperature, soil characteristics, and nutrient availability determine plant community composition and influence $\delta^{13}C$ and $\delta^{15}N$ at the base of the terrestrial food web (Ben-David and Flaherty 2012), and the isotope composition in organisms can change seasonally with food availability (e.g., fasting in winter can increase $\delta^{15}N$) (Hobson et al. 1993). These factors can differ both spatially and temporally resulting in variability of stable isotope signatures in terrestrial food webs. For example, the range of $\delta^{13}C$ and $\delta^{15}N$ in the vegetation–caribou–wolf food web, which has been studied extensively for biomagnification of organic contaminants (Kelly and Gobas 2001; Müller et al. 2011), is illustrative of the variation encountered in terrestrial food webs. In cottongrass (*Eriophorum vaginatum*), aquatic sedge (*Carex aquatilis*), and willow (*Salix pulchra*) from the same sampling sites, $\delta^{13}C$ varied widely and was only moderately enriched along the food web (1‰ to 2‰) (Müller et al. 2011). Lichen, caribou, and wolf had similar $\delta^{13}C$ values implying that the caribou were mainly feeding on lichen and the wolves mainly

on caribou or other lichen eating herbivores. In contrast, the $\delta^{15}N$ difference between lichen and caribou were rather large (7‰ to 8‰) compared to usually assumed $\delta^{15}N$ differences of 3.4‰ to 3.8‰, complicating the calculation of TL values used to calculate TMFs. Although additional source modeling (e.g., IsoSource) (Phillips et al. 2005) may be useful in understanding dietary contributions to isotopic mass balances, it is clear there are uncertainties that challenge the interpretation of stable isotope signals in terrestrial food webs, and additional study is needed before routine application for calculation of TMF values.

Visual gut content and fecal analysis have long been used in ecological and toxicological studies to determine the diet of organisms. These approaches rely on the identification and quantification of partially digested prey fragments and provide a snapshot of the diet at any given sampling time. They can be invasive (gut dissection) or noninvasive (collection of fecal samples or regurgitations). Although these procedures are inexpensive and do not require expensive instrumentation, they do have drawbacks. For example, the level of identification associated with gut content is sometimes limited by mastication and digestive processes that damage specimens resulting in fragments of tissue (Sample et al. 1993; Sample and Whitmore 1993). Furthermore, soft-bodied prey items may get digested more quickly than other items resulting in underestimation of these types of items in the diet. Considerable expertise in taxonomy is necessary to identify diet items based on tissue fragments, which makes the identification of specimens to species level difficult (Soininen et al. 2009). This can result in a somewhat subjective and even biased identification of specimens based on experience and professional judgment. Although fecal analysis is a noninvasive technique relative to analysis of gut content, such samples only contain fragments of tissue that were not digested and, thus, pose similar limitations and bias during identification. To avoid issues with digestion or partial digestion of items collected from gut content or fecal samples, predigestive samples can be obtained via throat ligature techniques, which have been successfully used for nestling birds (Mellott and Woods 1993; Powell 1984). This technique allows for an accurate determination of food items delivered to nestlings before digestion and can be used to ascertain site-specific concentrations of chemicals in food items.

Molecular methods can also be useful for determining diet and food web structure. With advances in DNA sequencing and polymerase chain reaction (PCR)-based technology, molecular methods have become more widely used by ecologists as tools for diet analysis. With the availability of free molecular databases, it is possible to use DNA barcoding (analyzing DNA-fragments) for organism identification even with short or degraded DNA sequences (Zaidi et al. 1999; Hajibabaei et al. 2006; Meusnier et al. 2008). Barcoding can be especially useful for species where the diet cannot be identified by gut-content analysis, observation, or other methods. Molecular methods are typically invasive in the case of gut dissections (Chen et al. 2000; Soininen et al. 2009; Carreon-Martinez et al. 2011) and noninvasive when using fecal samples (Corse et al. 2010; Zeale et al. 2011). PCR-based diet analysis is successfully accomplished in both aquatic and terrestrial systems and usually renders results with better taxonomic resolution than visual methods (Soininen et al. 2009; Carreon-Martinez et al. 2011). However, because each species may differ in the amount of DNA present per unit biomass and/or in tissue digestibility, molecular techniques provide merely a qualitative description of the diet (Zaidi et al. 1999). Laboratory testing may be used to calibrate PCR techniques for each food type to achieve semiquantitative results (Soininen et al. 2009; Deagle et al. 2010). One disadvantage of this approach however, is the inability to detect and distinguish primary and secondary predation (Sheppard et al. 2005). Because contamination of the sample with the predator's DNA is likely (e.g., during gut dissection), detection of cannibalism can also be difficult (Deagle et al. 2010; Carreon-Martinez et al. 2011). There are other practical considerations that may affect results such as gene and primer selection, sample preservation, temperature, and time since ingestion (for review, see Sheppard and Harwood 2005; King et al. 2008; Valentini et al. 2009).

Sample size. Sample size is a critical factor in statistical interpretation of bioaccumulation data, which is of great importance given the high variability and low sample sizes that are more the rule than the exception in field data sets. For example, results in Figure 2 indicate that a single BMF value should not be taken at face value without accounting for measurement variation. Forty-eight of the 59 (>80%) BMF and BSAF values in Figure 2 appear to be greater than 1 (indicating bioaccumulation potential). However, nearly 40% of these values were not statistically greater ($\alpha = 0.05$) than 1. Statistical comparison of field bioaccumulation information should be a requirement of all field studies reporting such information, and raw data, measures of variability and sample size should always be included to enable other researchers to use the data for assessing bioaccumulation potential (Borga et al. 2012; Conder et al. 2012).

The results shown in Figure 2 suggest that at least 3 or 4 replicates are the minimum sample size required for generating robust BMF data. Although avian BMFs in Figure 2B were variable (in part due to small sample sizes), higher values tended to offset variability and provide a statistically powerful estimate, capable of detecting significant bioaccumulation potential. Earthworm BSAFs for comparable chemicals tended to be 2 to 3 orders of magnitude lower than avian carnivore BMFs, but the large number of replicates ($n > 10$) improved statistical power (Figure 2A). Mammalian BMFs (Figure 2C) exhibited considerable variability, suggesting the need for greater replication than required to estimate avian BMFs.

Practical advice on obtaining samples

Obtaining adequate numbers of samples may be a practical challenge in terrestrial bioaccumulation studies. For example, in marine bioaccumulation studies, colonial breeding birds are often studied because collection of samples from animals is generally most efficient when they occur in aggregations or colonies. However, with some exception (e.g., European bee-eater, *Merops apiaster*, at a mining site) (Lopes et al. 2010) most terrestrial avian species generally do not aggregate. Some terrestrial studies take advantage of the willingness of cavity-nesting avian species to use nest boxes (e.g., tawny owl [*Strix aluco*] [Bustnes et al. 2007]; American kestrel [*Falco sparverius*] [Hebert et al. 1994]; tree swallow [*Tachycineta bicolor*] [Custer 2011]; small passerines *Parus* and *Ficedula* spp. [van den Steen et al. 2009; Berglund et al. 2012] house wrens [*Troglodytes aedon*] and eastern bluebirds [*Sialia sialis*] [Fredricks et al. 2010]; and the widespread European starling [*Sturnus vulgaris*] [Eens et al. 2013]). However, in

contrast to avian species, most mammalian species will generally not use provided shelter or nesting structures, which leaves the collection of mammalian samples more difficult.

A particularly efficient sampling approach involves leveraging sampling efforts from other activities or sampling programs. For example, tissue samples may be obtained from hunters (Conder and Lanno 1999; Müller et al. 2011) or from biologists and citizen scientists that collect animals found dead (van den Brink and Ma 1998). There are a growing number of environmental specimen banks located around the world (Becker et al. 2006) and long-term environmental monitoring projects which may offer the opportunity to provide tissue samples from controlled effort. Such specimen banks and programs have generally been designed to collect samples that are of value in assessing spatial and temporal variation in contaminant concentrations either in sentinel species and/or in species of particular conservation concern or interest (Elliott et al. 2005; Hebert and Weseloh 2006; Norstrom and Hebert 2006; Braune et al. 2007; Anderson et al. 2009; Helgason et al. 2009; Crosse et al. 2012). Data from these samples can be used to derive information on bioaccumulation potential or efforts to support modeling or data interpretation approaches. For example, patterns of PCBs, PBDEs, and stable isotopes of H, C, and N in eggs from peregrine falcons (*Falco peregrinus*) varied markedly among eggs collected from "big city" versus "coastal" nests, and revealed the need for data on dietary variation to decipher pathways and processes of biomagnification in a terrestrial top predators (Park et al. 2011). Archived liver samples of terrestrial raptors, such as Cooper's hawk (*Accipiter cooperi*), peregrine falcon, and Eurasian sparrowhawk (*A. nisus*), have been examined for spatial trends (urban to rural gradients) and accumulation patterns of POPs in relation to patterns of stable isotopes (Crosse et al. 2012, Crosse et al. 2013). When using archived samples from specimen banks, it should be made certain that concentrations of chemicals in the samples have not been affected by storage conditions and length of storage period. Especially in case of new emerging chemicals, without an analytical track-record issues with quality control and/or quality assurance should be considered when using archived samples.

Sublethal and minimally invasive sampling procedures should be considered for sampling programs, as lethal sampling may not be practical for many species, such as top predators, charismatic megafauna, and species of special conservation status (threatened and endangered). As long as species that are easily captured and handled in a manner that does not incur lethality, several types of tissues may be collected nondestructively (D'Have et al. 2006). For PFASs, blood is often used to determine trophic transfer based on the specific tissues (Tomy et al. 2004; Houde, Bujas et al. 2006). Relationships between levels of PFASs in avian feathers and liver have been established (Meyer et al. 2009). Feathers have also been used to monitor POPs in the chicks of predatory birds (Eulaers et al. 2011). Depending on the study purpose, the calibration of feathers, blood, and body tissues in target species is recommended particularly for less persistent contaminants that may occur at proportionally greater concentrations of these compounds than in other tissue types (Dauwe et al. 2005; Jaspers et al. 2007; Espín et al. 2010; Garcia-Fernandez et al. 2013). For mammals, hair can provide a noninvasive sample. For example, significant relationships have been established between concentrations of total PBDEs and PBDE congener-47 in the hair of hedgehogs (*Erinaceus europaeus*) relative to internal tissues, although less persistent congeners were more predominant in hair than other tissues (D'Havé et al. 2007). Preen oil has been used to analyze POPs in marine birds, and this may be applicable to terrestrial species as well (van den Brink et al. 1998).

Fecal matter is another noninvasive approach to monitor uptake and bioaccumulation in some mammalian species. For example, taking advantage of the use of regular latrine sites by river otters, a several researchers used scat samples to infer body burden of PCBs and other persistent contaminants (Mason et al. 1992; Smit et al. 1994; Elliott et al. 2008). Scats that have decreased relative levels of metabolizable PCB congeners reflect internal concentrations of otters and may therefore be used to assess accumulation in otters (van den Brink and Jansman 2006). The approach has been further developed using fecal DNA to identify individual animals and track their movement and contaminant exposure in time and space (Guertin et al. 2010).

CONCLUSIONS

Regulatory assessment of chemical bioaccumulation potential can benefit from data provided by terrestrial field studies, and initial work indicates that aquatic data may not completely represent bioaccumulation potential in terrestrial ecosystems. Data from samples obtained from terrestrial field studies can be used to derive bioaccumulation metrics that can be interpreted in existing chemical registration programs, and can also be used in the development, validation, and refinement of laboratory tests and models for prospective assessments of chemicals that have not been released to the environment.

In this article, we present practical recommendations and key issues that should be considered by scientists involved in research that elucidates chemical bioaccumulation potential in terrestrial systems and by regulatory authorities involved in the assessment of bioaccumulation potential within programs designed to register chemicals. BSAFs, BAFs, and BMFs appear to be the most suitable metrics that can be generated from terrestrial field studies. BSAF values are robust when based on measurements of soft-bodied soil organisms or plants. For higher trophic-level organisms, the BMF currently appears to be more robust that the TMF, which may reflect uncertainty in quantifying the trophic level of terrestrial animals using stable isotope signals. BMF values for lipophilic, nonpolar chemicals can be calculated using a variety of sample types (e.g., muscle, liver) if concentrations are expressed on a lipid-normalized basis. BMF values for avian carnivores appear to be particularly useful values for understanding chemical bioaccumulation, as this trophic guild appears to accumulate chemicals to a greater degree than other trophic levels, although this hypothesis deserves further investigation.

Sampling programs and efforts to evaluate data should strive for maximum efficiency in experimental design. It is essential, nonetheless, to achieve the appropriate statistical and interpretive power in field studies to optimize the achievement of scientific goals and, ultimately, the information needed to inform regulatory decision making. For some terrestrial food webs, as few as 4 to 5 predator–prey sample pairs may be sufficient for estimation of a relatively precise BMF. For other food webs, researchers may be able to take advantage of specimen tissue banks and long-term monitoring programs to reduce species collection efforts and obtain the tissues to support their research. Noninvasive and nonlethal sampling

efforts are also possible using samples of blood, hair, feather, feces, and other tissues.

More work is needed to improve the design of terrestrial field studies that address chemical bioaccumulation, as well as the subsequent application of field data to improve decision making in chemical registration programs. Several sources of uncertainty remain challenging such as seasonal variability in the diets of terrestrial organisms and the high spatial heterogeneity of the distribution of chemicals in terrestrial environments. The determination of food web relationships and diet preferences for predators is critical for developing BMFs and TMFs. Studies on the use of stable isotope ratios for quantifying trophic relationships in terrestrial ecosystems are needed to resolve uncertainties in the calculations of TMFs. In conclusion, addressing recommendations provided in this overview, as well as future scientific and regulatory discourse, will facilitate the application of terrestrial field data as a line of evidence in the assessment of chemical bioaccumulation potential.

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SUPPLEMENTAL DATA

Data S1. Specific aspects of environmental fate and behavior of compounds in terrestrial ecosystems

Data S2. Accumulation metrics (BSAF, BMF)

Data S3. Terrestrial field studies on bioaccumulation potential of chemicals

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Use of terrestrial field studies in the derivation of bioaccumulation potential of chemicals

SUPPLEMENTAL INFORMATION

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SI1 Specific aspects of environmental fate and behaviour of compounds in terrestrial ecosystems

Bioavailability

Soil properties (e.g., organic carbon content and quality, water-holding capacity, pH and cation-exchange capacity) affect the availability of compounds for uptake by organisms (Amorim et al., 2005; Chung and Alexander, 2002; Cornelissen et al., 2005). Furthermore, the availability of organic compounds in soil may decrease in time due to ageing (Belfroid et al., 1995). Toxicity of pyrene for instance, decreased for collembolans (*Folsomia candida*) with increasing soil organic matter content and with ageing (Styrishave et al., 2008). Chemical properties of organic compounds are important factors that drive chemical uptake by soil dwelling organisms (Gevao et al., 2001); (De Silva et al., 2010; Garcia-Santos and Keller-Forrer, 2011). Methods used to predict bioaccumulation potential of organics in aquatic systems rely on measures of hydrophobicity (i.e., logKow). However, in terrestrial systems, log Kow alone does not explain or predict bioaccumulation (Belfroid et al., 1995).

Not all organic compounds follow similar fate patterns in terrestrial ecosystems. For example, nitroaromatics and nitramines (used as explosives, energetics, propellants) are typically reduced to amines by microbes in soils and can become irreversibly bound to the organic material, resulting in non-extractable residues (Johnson et al., 2009). Nevertheless, dermal exposure to bioavailable nitroaromatics can be significant and more important than oral routes in some species (e.g., amphibians) (Johnson et al., 1999). Dermal uptake is also a significant route of uptake in earthworms (Henson-Ramsey et al. 2009).

Biotransformation and degradation

Persistence of a chemical in the environment is determined by its resistance to physicochemical and biological degradation. The dissipation of organic compounds from soils in terrestrial systems results from mechanisms such as microbial degradation, chemical hydrolysis, photolysis, volatility, leaching, and surface runoff. In addition to soil properties (e.g., pH, OM, biomass, redox status), environmental conditions (e.g., temperature and moisture) and physicochemical properties of organic chemicals are drivers of these processes (Racke et al., 1996; Singh et al., 2003). For instance, some organophosphorus and carbamate insecticides (terbufos, phorate, isofenphos, aldicarb) may degrade in aerobic conditions (mainly through oxidation reactions), while other pesticides, including organochlorine insecticides with free nitro- groups (e.g., parathion, fenitrothion, PCNB, chlomethoxynil) degrade under anaerobic

conditions (Adhya et al., 1987; Racke et al., 1997). Additionally, 2,4,6-trinitrotoluene is demethylated to trinitrobenzene in aerobic environments; however, nitro groups are reduced by microorganisms to mono-amino derivatives in wet soils (Johnson et al., 2009; Monteil-Rivera et al., 2009). Vapour pressure, water solubility and partition and adsorption coefficients of compounds are often the principal parameters affecting dissipation mechanisms of chemicals in soil (Kromer et al., 2004). While partitioning properties drive biomagnification of non-polar organic chemicals in aquatic food webs, biotransformation seems to have more profound effects on bioaccumulation and biomagnification in terrestrial systems (Kelly et al., 2007; McLachlan et al., 2011). Compounds that are readily biotransformed to water soluble metabolites exhibit the lowest potential to biomagnify in food webs, though exceptions exist (Johnson et al., 2009; Blankenship et al., 2005; van den Brink and Bosveld, 2005; Voorspoels et al., 2007). Enzymatic activities differ among tissues which may affect bioaccumulation. For example, in the tiger salamanders (*Ambystoma tigrinum*), skin was primarily responsible for the reduction of 2,4,6-trinitrotoluene *in vivo,* either through dermal metabolism or microbial activity (Johnson et al., 2000). The skin however, showed limited phase II metabolic activity. These metabolic differences resulted in diverse profiles of metabolites among the different tissues, potentially affecting uptake patterns of the parent compound.

SI2. Accumulation metrics (BSAF, BMF)

Data Assessment

We used the following metrics to compare bioaccumulation data generated from terrestrial field studies with aquatic field studies (e.g., Figure 1 in the primary paper): BCFs, BAFs, aquatic biota-sediment accumulation factors (BSAFs) for aquatic invertebrates; terrestrial biota-soil accumulation factors (BSAF) for plants and invertebrates; and biomagnification factors (BMFs) for fish and for marine and terrestrial birds and mammals (Kelly and Gobas, 2001; Müller et al., 2011; Voorspoels et al., 2007; Winter and Streit, 1992).

BMFs were the ratio between lipid-normalized, sample mean concentration of predators and lipid-normalized sample mean concentrations in prey. BSAFs were calculated by dividing lipid-normalized concentrations of chemical by organic carbon-normalized concentrations in soil or sediment for invertebrates, and wet weight or dry weight concentrations in plant tissue or soil for plant BSAFs. For pyrene, lipid and organic carbon normalized values were not available and data are reported in wet and dry weights for BSAFs. BSAFs in sediment and soil BMFs for PFOS were calculated from concentrations in tissues expressed on a wet weight basis,

because PFOS does not partition to lipids (Conder et al., 2008). BCFs and BAFs values for PFOS are based on dividing concentrations in tissue on a wet weight basis by concentrations in soil.

Statistical analyses

Values were combined by tissue type and chemical for particular predator-prey relationships, and in some cases, BMF values were combined from multiple areas, seasons, and even studies. Where BMF values ranged over more than one order of magnitude, values were $\log_{10}$-transformed to meet statistical test assumptions. Mean concentrations in predator and prey from Voorspoels et al. (2007) were presented for multiple tissue types without a measure of variance. To propagate an error term, we assumed the two samples from different tissue types (liver and muscle) could be considered replicate measurements of the steady state concentrations in the lipid in the animal. Concentrations of chemicals in the liver of predators were divided by concentrations in the liver of prey and the resultant lipid-normalized BMF was averaged with that from the same mathematical operation with matched predator/prey concentrations in muscle. For the assessment as to whether field-derived BSAFs or BMFs differed from the threshold value for bioaccumulation potential (1), a simple confidence interval (CI) approach was used (Borgå et al., 2012). First, the 95% confidence intervals (CIs) were calculated. The lower 95% CI was used as a threshold; if the lower 95% CI was below 1, the average was not considered to be different to 1 and the data were interpreted as not exceeding the bioaccumulation potential threshold. This approach is directly analogous to a simple *t*-test comparison of sample data to a fixed threshold value (i.e., 1).

Evaluation of BMFs Based on Different Tissue Types

We compiled BMF values from key papers that included concentrations of organic pollutants in both liver and muscle of terrestrial food webs, calculating matched BMF_{Liver} and BMF_{Muscle} (Figure 3 in the primary paper). Statistical significance of the compiled data shown in Figure 3 of the primary paper was assessed, using 95% CIs on BMF_{Liver} and BMF_{Muscle}. In 23% of the cases, both BMF_{Liver} and BMF_{Muscle} were statistically greater than 1 and in 2% both BMFs were statistically lower than 1 (Figure SI-1A). In 71% of the cases, at least one tissue BMF was not significantly different from 1 ($\alpha = 0.05$), indicating inconclusive results a majority of the time. In most of those cases (69%) both BMF_{Liver} and BMF_{Muscle} were not significantly different from 1, reflecting high variability in the terrestrial BMF datasets, and a crucial need to evaluate data statistically in bioaccumulation studies. In cases of disagreement (those 31% of the pairs where

at least 1 BMF was not different than 1), no preferential tissue could be selected to indicate a BMF that was statistically greater than 1.

In contrast, if statistical analysis is omitted from an interpretation of the results, 63% of the cases (instead of 23%) indicate that both BMF_{Liver} and BMF_{Muscle} values are greater than 1 (Figure SI-1B). Additionally, review of the data without a statistical analysis indicate that for 10% of the cases (instead of 2%), both BMF_{Liver} and BMF_{Muscle} values are less than 1 (Figure SI-1B). Thus, omitting a statistical comparison of BMFs to the value of 1 provides a false confidence in the results and can lead to errors in identifying bioaccumulation potential.

As noted in the primary paper, we also compiled BMF_{Liver} and BMF_{Muscle} values for perfluorinated compounds, as shown in Figure SI-2. BMF_{Muscle} showed the better agreement with $BMF_{Wholebody}$ (Figure SI-2B) than did BMF_{Liver} for both wolves and caribou (Figure SI-2A).

Overall, the results in Figure SI-2 show the importance of statistical evaluation of BMF. It also highlights that neither BMF_{Liver} nor BMF_{Muscle} appear to be more reliable in assessing bioaccumulation. BMF_{Muscle} and BMF_{Liver} agree with respect to bioaccumulation assessment when sample size is large enough to determine statistical significance and when samples are lipid normalized. While additional research with more tissues is needed, these initial results suggest lipid normalised concentrations of non-polar organic contaminants are likely to be adequate for terrestrial bioaccumulation assessment. However, in case of other types of contaminants, the applicability of specific sample types may need to be validated before application in B-assessment.

SI3. Terrestrial field studies on bioaccumulation potential of chemicals

There is a wealth of available field data that can be used to evaluate bioaccumulation potential of substances that have been released in the environment, and can inform model and laboratory approaches used to predict bioaccumulation potential of compounds currently under review. In a classic study, concentrations of DDTs were monitored in soil, vegetation, insects and higher vertebrates collected from an agricultural drainage following aerial application of DDT (Rudd et al., 1981). Residues initially spiked in both herbivorous and carnivorous species, yet this response was followed by a protracted and "unexpected trophic increase" in carnivorous arthropods (*Arachnida* spp., *Coleoptera* spp.) and the loggerhead shrike (*Lanius ludovicianus*). More recent work conducted in British Columbia and Ontario, Canada (Harris et al., 2000; Hebert et al., 1994) and Schlüchtern, Germany (Winter and Streit, 1992) provide compelling

evidence of organochlorine pesticide (principally DDT and metabolites) and PCB biomagnification in terrestrial food webs (soil to earthworms to eggs or nestlings of invertivorous species of birds). Statistical differences, relationships and trends in concentrations were apparent among trophic levels in those robust datasets; however BMFs or TMFs were not estimated. For more contemporary groups of pesticides (e.g., organophosphorus insecticides, second-generation anticoagulant rodenticides) and some pharmaceuticals (e.g., diclofenac, a non-steroidal anti-inflammatory drug), there are extensive exposure data and toxicological evidence of terrestrial food chain and food web transfer (Cobb et al., 2000; Eason et al., 2002; Oaks et al., 2004; Taggart et al., 2007). However, efforts have not been undertaken to estimate trophic transfer factors for these newer compounds.

Many datasets are available for terrestrial hazardous waste sites (e.g., Superfund Sites reviewed in Sheffield et al. 2001; Rocky Mountain Arsenal, Colorado (Edson et al., 2011)), however their primary focus was not on quantitatively describing movement of organic contaminants through food webs. A notable exception was a study on a terrestrial food web at impoundments of the Kalamazoo River flood plain in Michigan (Blankenship et al., 2005). Soils, plants, invertebrates, birds (eggs, nestling and adults), and small mammals were collected and quantified for various PCB congeners. Concentrations were often greatest in soils compared to invertebrate, avian and mammalian biota (lipid normalised biota-soil accumulation factors <1). However, biomagnification was apparent from low level to higher trophic level biota (e.g., terrestrial invertebrates to shrew BMF: 16; small mammals to great horned owls eggs BMF: 13).

In a study of PBDE congeners in three terrestrial food chains in Belgium (Voorspoels et al., 2007), lipid normalised BMFs ranged from 2 to 34 in both food chains leading to avian predators, and the BMFs were related to log K_{OW} of the compounds. Biomagnification appeared absent in the small mammals to fox food chain, which was attributed to the greater ability of fox to biotransform xenobiotics when compared to birds. Another study examined PBDE congeners in insects, brown rat (*Rattus norvegicus*), Eurasian tree sparrow (*Passer montanus*) and common kestrel (*Falco tinnunculus* (Yu et al., 2011)). BMFs varied from 1 to 160 for individual congeners, and from 1.6 to 6.9 for total PBDEs. In contrast to the study of Voorspoels and coworkers (2007), BMFs were not related to log K_{OW}, and lower brominated BDE congeners 47, 99 and 100 did not biomagnify, possibly due to metabolism. BDE153 exhibited the greatest potential for biomagnification (Yu et al., 2011). Additional studies that were undertaken in three passerines found in South China revealed significant relationships between

$\delta^{15}N$ and concentrations of the larger BDE congeners 186 and 209, and decabromodiphenyl ethane in three passerine species (Sun et al., 2012).

Persistent organochlorine pesticides, PCBs, and perfluoroalkyl substances (PFAS) have been studied in the lichen-caribou-wolf food chain in the central and western Canadian Arctic (Kelly and Gobas, 2001; Müller et al., 2011). This food web is unique in that it receives anthropogenic pollutants solely from atmospheric deposition and feeding relationships are well-known. HCB, β-HCH, PCB153 concentrations increased significantly with increasing trophic level. However, concentrations of PCB52 did not increase significantly between trophic levels, suggesting elimination/metabolism of this congener. The most prevalent PCB congeners in lichens consist of lower chlorinated compounds such as CB52, 66/95, and 101; caribou accumulated those congeners, in addition to various higher chlorinated congeners such as PCB118, 138, 153, and 180 with PCB153 predominating. Concentrations of p,p'-DDE, a compound known to bioaccumulate in aquatic food webs, declined significantly in this terrestrial food chain, also indicating biotransformation in both prey (caribou, *Rangifer tarandus groenlandicus*) and predator (wolf, *Canis lupus*). BMFs of β-HCH ranged from 3.4 to 28 for wolves, while no substantial biomagnification or trophic dilution of α-HCH was observed (average TMF = 0.54).

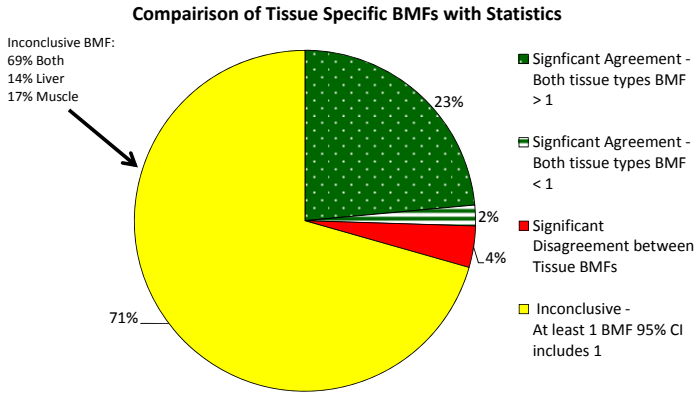
The lichen/vegetation-caribou-wolf food chain was also studied to assess bioaccumulation of PFASs (Müller et al., 2011). Low concentrations of C8 to C12-perfluorocarboxylates (PFCAs) along with PFOS were found in vegetation. Wolf liver showed highest concentrations, followed by caribou. Signatures of $\delta^{13}C$ and $\delta^{15}N$ showed that the caribou were feeding mainly on lichen and wolves on caribou. TMFs were highest for PFCAs with nine to eleven carbons (TMF = 2.1-2.9) as well as PFOS (TMF = 2.3-2.8). Calculation of the TMF in this terrestrial food web was challenging due to seasonal variation in caribou diet and to the large difference between lichen and caribou compared to usually assumed trophic enrichment ($\delta^{15}N$) of 3.4-3.8‰. The relationship of TMFs for PFCAs and PFOS with the chain length in the terrestrial food chain was similar to previous studies for Arctic marine mammal food web, but the absolute values of TMFs were about two times lower for this study than in the marine environment. Magnification of current use pesticide in this food chain was limited, with BMFs below 1, due to the fact that mammals can metabolise the compounds (Morris et al. 2014).

Contamination of a terrestrial food web by guano from a northern fulmar colony was studied on Devon Island in the Canadian arctic (Choy et al., 2010). Concentrations of ΣDDT, ΣPCBs and CB153 were highest in snow buntings (*Plectrophenax nivalis*) (64-168 ng g^{-1} ww) followed

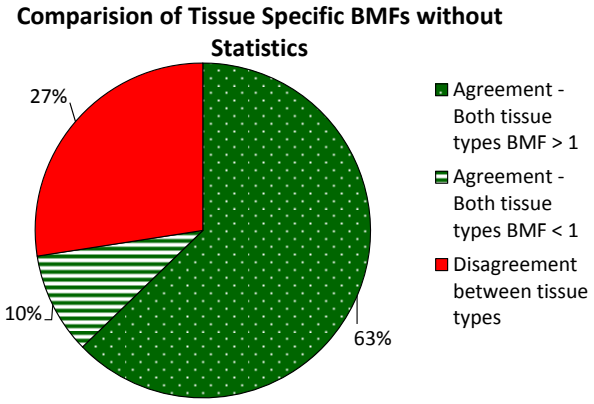
by ermine (18-54 ng g^{-1} ww), lichen (*Xanthoria elegans*) (0.3-3.3 ng g^{-1} ww) and lemmings (0.11-0.76 ng g^{-1} ww). There was an exponential relationship between log PCB concetrations [ng g^{-1} ww] and trophic level (measured with $\delta^{15}N$ values) for PCBs and ΣDDT in this food web**.** However, no TMF values were reported.

Bioaccumulation of several persistent organic pollutants was examined in detail in the air to soil/plant to cow to human agricultural food chain (McLachlan 1996; Czub and McLachlan 2004; McLachlan et al. 2011). For moderately hydrophobic contaminants with low volatility (e.g., lower chlorinated biphenyls), concentrations remained relatively constant at different levels of agricultural food chains, with some biomagnification occurring in the cow. However, concentrations of volatile and very hydrophobic compounds decreased in agricultural food chains (biodilution) due to i) ineffective accumulation from air to vegetation, ii) poor absorption by the digestive tract, and iii) metabolism of some compounds by domestic farm animals and humans. Compounds with highest potential for bioaccumulation were those with a log K_{OW} between 2 and 9 and a log K_{OA} between 6 and 10. Notably, persistence in the environment and food web were more significant factors to human exposure than partitioning properties. Modelled data for a range of chemicals with disparate physico-chemical and partitioning properties for humans, resulted in merely an order of magnitude difference in bioaccumulation factors (varying between 10 and 120), while biotransformation rate constants (ranging over 6 orders of magnitude) had a more dramatic effect on BMFs (McLachlan et al., 2011).

Relationships between **$BMF_{Wholebody}$ and BMF_{Liver}** ***and*** **$BMF_{Wholebody}$ and BMF_{Muscle}**

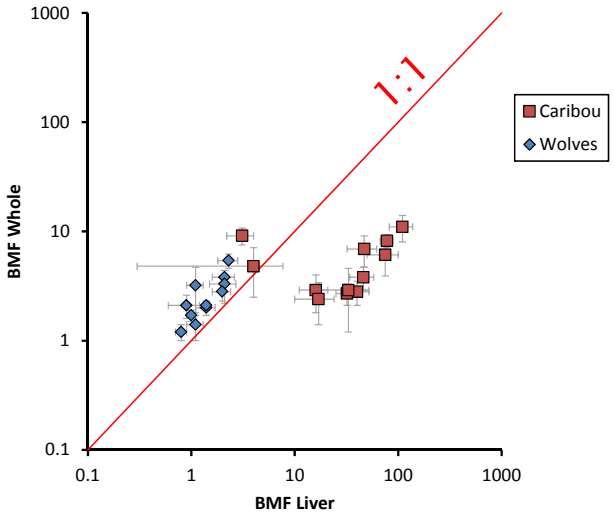


A

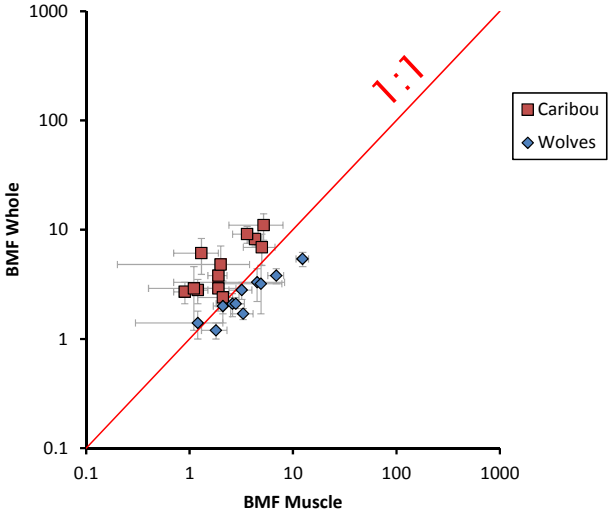


B

Figure SI-1. Comparison of B-assessment results (a) with and (b) without statistical consideration (i.e. null hypothesis that BMF values equal 1).



A



B

Figure SI-2. A Correlation between $BMF_{Wholebody}$ and BMF_{Liver}. B Correlation between $BMF_{Wholebody}$ and BMF_{Muscle}. Error bars represent 1 standard deviation of the mean. Red line indicates a 1:1 relationship.data: (Houde et al., 2006; Müller et al., 2011)

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