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Biomagnification in marine systems: the perspective of an ecologist

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Abstract

Biomagnification is the process where xenobiotic substances are transferred from food to an organism resulting in higher concentrations compared with the source. It is widely believed that this is a general phenomenon for marine food webs. An analysis of 148 papers with biomagnification in the title shows that under half show biomagnification. Of studies on metals only organic mercury shows biomagnification and most metals are regulated and excreted and do not biomagnify. Of the studies on organic compounds 67% claimed to show biomagnification. However, bioconcentration (uptake from the surrounding water) is the most usual way that organic compounds are accumulated in organisms from invertebrates to and including fish. Only in sea-birds and marine mammals is food intake the major route and where biomagnification can be clearly shown. Body concentrations of organic compounds vary with lipid content and thus in order to compare across species normalisation to uniform lipid content should be done. Yet often this is not done so data purporting to show biomagnification merely relate to differing lipid content in the different species studied. Finally suggestions are made as to how data can be collected to better interpret the process of biomagnification in marine food webs.

Keywords: Food webs; Metals; Organic chemicals; Bioconcentration; Biomagnification

1. Introduction

Rachel Carson's book "Silent Spring" started an era of research into biomagnification up food chains. Generations of students, myself included, were brought up knowing top predators, such as hawks and eagles, had accumulated so much dichlorodiphenyltrichloroethane (DDT) that eggs laid had shells so thin that chicks died and ultimately populations were affected. Following the lessons from this book it is widely assumed that biomagnification up food webs is a general rule in all ecological systems. Yet in the 1970s Isaacs (1973) suggested that this was not the case in marine systems as marine food chains were more open than terrestrial ones. Isaacs postulated that terrestrial systems were characterised by tight predator–prey relationships where only one or at most a few species of prey were consumed. In marine systems predators consumed a wide range of prey that were smaller than the predator. As contaminant concentrations vary between species and among age groups within species there was less likelihood of biomagnification in marine food chains.

Biomagnification is usually defined as the transfer of a xenobiotic chemical from food to an organism, resulting in a generally higher concentration within the organism than source, (Connell, 1989, 1990; Rand et al., 1995). However, not all authors use this definition and some define biomagnification as the increase in concentration between trophic levels, if the biomagnification factor (concentration in predator/concentration in prey) >1, then the compound is biomagnified. The problem with this definition is that the mechanism leading to the increase may be simply due to accumulation from the surrounding water whereas Connell's definition restricts the term to concentration increases that result from food intake alone. I believe that restricting the definition solely to food intake is preferable as the mechanism of uptake is defined and follow this common usage of the term.

Yet it is not only via food intake that contaminants can accumulate within organisms, (McKay and Fraser, 2000). There are two other mechanisms for uptake of contaminants by biota, bioconcentration and bioaccumulation. Here I follow the definitions given by Rand et al. (1995) where, bioconcentration is defined as the

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uptake of a chemical by an organism directly from the abiotic environment resulting in a higher concentration within the organism and bioaccumulation is defined as the uptake of a chemical by an organism from the abiotic and/or biotic (food) environment, that is from all sources. Once contaminants are within an organism they may be metabolised and/or excreted so that the concentration is a balance between intake, by whatever means, and regulation.

Since the 1970s we have obtained much more data from a variety of systems and have developed new techniques to assess feeding relationships in ecosystems. Here I review these data from an ecologists perspective and assess the implications for generalisation concerning biomagnification.

2. Food chains and food webs

In his 1970 study Isaacs used the ratio of a non-essential element caesium (Cs) to an essential element, potassium (K), Cs/K, to examine the trophic structure of the food web off the open coast of California compared this with that of the enclosed Salton Sea. Table 1 shows the results.

For the same species within each system it is clear that there is bioaccumulation up the closed system of the Salton Sea, but not in the open sea. Isaacs suggested that the open coast was an unstructured system where predators fed on a variety of prey so that contaminants such as caesium were not biomagnified up the food chain. In the enclosed sea predators had a more limited choice of prey and thus any contaminants would biomagnify up the food chain. Further analyses by Young et al. (1981, 1987) and Young (1984) showed similar patterns using this method. However, other data using the Cs/K ratio, on DDT in species at different trophic levels in St Margarets Bay, Canada (Hargrave and Phillips, 1976) and on mercury in a Canadian estuary (Williams and Weiss, 1973) showed that there was in fact no biomagnification. Yet apart from Young's studies further work using the Cs/K ratio has not been done.

Instead of the Cs/K ratio, recently attention has focussed on the use of δ^{15}N as an indicator of food web structure (Peterson and Fry, 1987). It was shown that the trophic hierarchy of organisms taken from systems with unknown feeding relationship could be determined using the δ^{15}N content since the δ^{15}N content is increased by $\approx$3‰ as the next higher trophic level is sampled (Broman et al., 1992). Applying this technique many studies show that contaminants are often significantly correlated with the δ^{15}N content (Kidd et al., 1995; Broman et al., 1992). Given the correlation it is assumed that biomagnification is the cause and that contaminant levels increase at higher trophic levels due to the food intake. But is biomagnification a general rule and is food intake the only mechanism for obtaining greater concentrations at high trophic levels? It could be argued, using Isaacs (1975) study as a basis, that in many freshwater systems the food webs are highly structured since the predators can consume all the prey in a pond or lake and thus any contaminants will biomagnify. Thus there may be real differences in the structure of freshwater and marine systems, which account for differences in apparent biomagnification between the two systems. So what is the evidence that biomagnification occurs at all?

Table 1
Structure of marine food chains as measured by the caesium/potassium ratio, Cs/K (from Isaacs, 1975)

Organism	Salton sea	Gulf of California
Water	3.5 ± 0.1	0.82 ± 0.05
Pooled invertebrates	3.8 ± 8.5	
Mullet (*Mugil cephalus*)	9.0 ± 1.1	15.2 ± 1.9
Sargo-Croaker (*Urbina roncador*)	26.0 ± 2.7	13.2 ± 1.0
Corvina (*Cynoscion parvipinnus*)	57.7 ± 7.7	10.4 ± 0.8

3. Data analysis

A total of 325 papers were analysed in the Aquatic Sciences and Fisheries Abstracts (ASFA) database, published between 1970 and 2000 where data were presented and the word biomagnification occurred in the title. Of these 148 papers presented data (references can be found at http:/www.uio.no/~johnsg) and were examined. A total of 86 were used for detailed analyses 51 were marine and 35 freshwater papers. The results show that biomagnification was reported in 42% of the papers, biomagnification was absent in 47% and possible biomagnification was reported in 11%.

One major problem with analysing all the data is that the methods used varied widely from study to study. One common occurrence was that analyses were done using whole organisms at the bottom of the food chain and specific tissues at the top. The reason for this is that at the bottom of food chains the organisms are usually small in size and thus are analysed as total body content, whereas in marine mammals tissues such as liver and muscle are often used. It is well known that contaminants are often stored in special organs such as the liver or are excreted through special structures, such as feathers in birds. Bird feathers will therefore, have high levels of contaminants up to the moulting period and thereafter may have relatively low concentrations until contaminants are stored again. Thus comparisons of for example, whole body tissues in polychaetes with liver levels in a consumer of polychaetes will give misleading results on biomagnification, higher concentrations in the liver tissue may simply reflect that this is where

Table 2
Analysis of biomagnification in relation to the compounds analysed, from papers in the ASFA database (see text for details)

Compound	Number of papers	% Showing biomagnification
Mercury (Hg)	14	42
Other metals	14	0
PCB	35	54
Tributyl tin (TBT), Butyl tin (BT)	7	28
DDT and DDE	6	67
PAHs	2	0

xenobiotic chemicals are stored and have nothing to do with the process of biomagnification. A recent example of such a study is that of an Antarctic ecosystem (Bargagli et al., 1998), which purports to show biomagni-fication yet the tissues examined were quite different between trophic levels, whole body tissues in lower organisms, muscles and livers in higher organisms and feathers in sea-birds. Compared with other tissues marine birds feathers usually have very high concentrations (Monteiro et al., 1998). One must expect, therefore, that the liver and bird feathers will have higher concentrations of contaminants without this necessarily having anything to do with accumulation by means of a food web.

Despite such shortcomings there are some major patterns that are apparent when one analyses for biomagnification among different contaminants. Table 2 shows that with organic contaminants biomagnification occurred more frequently than for heavy metals. For Polychlorinated biphenyl's (PCBs), and DDT biomagnification occurred in over 50% of the cases studied (54% and 67% respectively). Polychlorinated aromatic hydrocarbons (PAHs) probably do not show biomagnification.

With the exception of mercury, there is little evidence of the biomagnification of metals. Mercury shows approximately the same frequency of biomagnification, ≈42% as the organic chemicals studied. Yet a breakdown of these studies show that methyl mercury is particularly liable to biomagnify (Riisgaard and Hansen, 1990; Reinfelder et al., 1998), whereas inorganic mercury does not. Here I examine first the uptake and metabolism of metals and then of organic contaminants.

4. Metal uptake

Goodyear and McNeill (1999) have reviewed uptake rates of metals by freshwater macro-invertebrates. They note that zinc (Zn), copper (Cu), lead (Pb) and cadmium (Cd) are the most frequently studied. They considered three feeding guilds collector-gatherers, scraper-grazers and predators and examined uptake rates from water (soluble form) and sediment (particulate form) for pooled data. They found that animals took up Zn in direct proportion to levels found in sediments, with predators having the highest concentrations. However, there was only a slight increase in concentration within animals compared with water values suggesting regulation occurs. With Cu there were similar uptake rates from the water and sediment and rates were similar across feeding guilds indicating that biomagnification did not occur. Pb was taken up in a similar way to copper from both water and sediment, but there was a suggestion that there may be biomagnification since the feeding groups responded differently. Cd concentrations were low and trends difficult to interpret. Thus from this comprehensive survey again biomagnification does not appear to be the rule, it is rather the exception. Whereas Cd was difficult to interpret in this study Bargagli et al. (1998) have studied concentrations in a food web in the Mediterranean Sea and find that at high trophic levels Cd concentrations are lower than at the bottom of the food chain. There is no evidence of biomagnification of Cd in this marine food chain.

In a study of Hg in Swedish lakes Parkman and Meili (1993) showed that concentrations were highest in three detritivores and predators had significantly lower concentrations. Thus there was no evidence of biomagnification. Atwell et al. (1998) studied Hg in an Arctic system and related food web Hg levels to δ^{15}N values. They measured concentrations in whole body tissues in invertebrates and in muscle tissue in other organisms so the comparisons are not strictly valid. Yet they found that polar bears (*Ursus maritimus*) had lower concentrations than ringed seal (*Phoca hispida*) on which they feed. In addition whereas marine birds and mammals had the same δ^{15}N values Hg content varied widely. A simple explanation based on biomagnification cannot be used here and bioconcentration, excretion and other factors such as varying lipid concentrations between species (see later) need to be considered.

Riisgaard and Hansen (1990) did experiments transferring blue mussels (*Mytilus edulis*) from an uncontaminated area to a Hg-contaminated area. Organic Hg was taken up at much faster rates than inorganic Hg, yet flounders (*Platichthys flesus*) transferred under the same conditions did not take up Hg. By force-feeding flounders with Hg-contaminated food they were able to show that inorganic Hg accumulated in the kidneys and liver but was excreted. Organic Hg on the other hand was transformed to inorganic Hg and excreted but some was stored in muscle where it accumulated with fish age. Thus from these studies it is clear that organic Hg may biomagnify and accumulate in muscle tissue, but that inorganic Hg is taken up by bioconcentration rather than biomagnification and can be excreted.

Organic Hg accumulated to high levels in feathers taken from sea-birds off the Azores (Monteiro et al., 1998). In this study there were no trophic level differences as the birds all fed on squid or fish. Overall there

was a large increase in concentration from food to feathers. The species with the highest Hg levels fed on mesopelagic fish and squid whereas those feeding on epipelagic fish and squid had lower Hg levels in their feathers. It is because the feathers are sites of organic Hg excretion that high levels are found.

Reinfelder et al. (1998) have developed a simple model that predicts biomagnification with metal uptake. They use two factors in the model an elimination rate constant and the assimilation efficiency. They plot data for a variety of marine organisms (copepods, bivalves, other invertebrates and fish) and show a convincing curvilinear relationship. Only methyl mercury (CH_3Hg) is liable to biomagnify according to their model as it has low elimination rates and high assimilation rates in the organisms studied. Other metals are not expected to biomagnify. This model gives a convincing explanation for the observed patterns of metal biomagnification in marine systems.

5. Organochlorines and biomagnification

Table 2 suggests that for DDT and PCBs biomagnification occurs, in half to two-thirds of the data sets. Yet high concentrations may not be due to biomagnification (uptake through food), as organisms may take up contaminants through their body surface or respiratory organs by diffusion, which is the process of bioconcentration. For most small organisms such as plankton, polychaetes, bivalves and crustaceans the major route of intake is by respiratory surfaces. Randall et al. (1998) have shown in experiments that in trout (*Salmo trutta*), by far the largest proportion of tetrachlorobenzene was taken up through the gills. They concluded that intake from food was only a relatively minor pathway. If this is generally true then there is no reason to assume, at least with organisms up to fish that biomagnification is the only explanation for the finding of higher concentrations at higher trophic levels.

Kidd et al. (1995) examined toxaphene concentrations in a Canadian Arctic lake. The data showed that toxaphene concentrations increased with $\delta^{15}N$ content, which they claimed was due to biomagnification. The data are however, correlative and do not necessarily indicate biomagnification as other explanations, such as bioconcentration are possible.

Data are available on concentrations of a variety of organochlorine compounds in the fauna of the semi-enclosed Bothnian Bay and the more open Bothnian Sea in the Baltic Sea, (Strandberg et al., 1998). Hexachlorocyclohexanes (HCHs) do not show evidence of biomagnification whereas Hexachlorbenzenes (HCBs), DDTs, Chlordanes (CHLs), dieldrin and PCB all show higher values in the predator than the prey. Surprisingly the more open Bothnian Sea has higher values than the Bothnian Bay, which suggests that sources come from the open Baltic Sea rather than land-based sources within the Bothnian Bay. The data however, were obtained from analysis of whole organisms in zooplankton and mysids whereas no information is given on what part of the fish was analysed. The data were standardised for lipid however (see below). Thus again biomagnification is assumed without there being any exploration of other mechanisms by which organisms may take in organochlorine chemicals.

Data from PCBs in a lake food web (Paterson et al., 1998) showed that PCB concentrations across the whole data set did not increase from plankton to lake trout (*Salmo trutta*). There were however, differences between two species of fish with cisco (*Coregonus artedi*) having higher PCB concentrations than walleye (*Stizostedion vitreum*). The reason is related to the differences in lipid concentrations walleye having only 1.7% lipid whereas cisco had 2.9%. Since most organochlorine compounds are fat soluble species with higher lipid content will have higher concentrations of PCB. Normalisation of data to unit lipid concentration should clearly be common practice, but many data sets do not use such procedures and naively interpret results as evidence for biomagnification. Paterson et al.'s (1998) data also showed another common finding that highly chlorinated PCBs show higher concentrations in fish than zooplankton suggesting that in fact these compounds could be biomagnified, whereas the other compounds were not.

In a study of mirex and PCBs in beluga whales (*Delphinapterus leucas*) in Canada (Muir et al., 1996) biomagnification was measured simply as the concentrations in blubber compared with food on a lipid basis. Although biomagnification appears to occur there were no consistent patterns as blubber tissue was $\approx$30 times that of tom cod (*Microgadus tomcod*) but only 15 times that in eel (*Anguilla rostrata*) and smelt (*Osmerus mordax*) in the St Lawrence estuary. In the Arctic beluga whales blubber had 50 times higher PCB concentrations than arctic cod (*Boreogadus saida*) but only 30 times higher than in zooplankton. Again biomagnification is assumed without any consideration of alternative hypotheses or mechanisms.

6. Mechanisms of bioaccumulation, bioconcentration and biomagnification

Fish accumulate higher concentrations of DDT, dieldrin and endrin than are found in the surrounding water and in invertebrates living in the water due to passive partitioning of the chemical between the aqueous environment and the organic compartment (Leblanc, 1995). DDT concentrations in the organisms may accumulate to many thousand times that of the water by this route. Many studies have shown that uptake of

organic contaminants relates to the lipophilicity of the chemical. The fugacity (the propensity of a chemical to leave the compartment it is associated with for another compartment, Leblanc, 1995) is high for lipophilic chemicals in aqueous environments to organisms. Fugacity can be estimated rather accurately by using the octanol:water partition coefficient. Thus the ratio of the steady-state concentration between organism and environment correlates well with the octanol–water partition coefficient. Even in sediment-living invertebrates, which ingest large amounts of sediment, the major pathway for intake of contaminants appears to be from the surrounding pore-water rather than food.

As lipophilicity is a major factor, the amount of lipids within organisms is a major determinant of organic contaminant concentration. Leblanc (1995) has analysed the PCB content of various organisms from Lake Ontario and shows a direct correlation with trophic level and with lipid concentration. There is therefore, no need to invoke biomagnification as an explanation for the results obtained, it is simply that at higher trophic levels organisms tend to have a higher lipid concentration.

However, Vander Zanden and Rasmussen (1996) showed that in a study of the fauna of 21 lakes in Ontario, Canada that although PCB levels were associated with increased lipid at higher trophic levels normalising to a standard lipid there was still an increase in PCB with trophic level suggesting biomagnification. Thus analyses of contaminant levels in different organisms must be normalised to the lipid content. Whilst this is the practice today this was not so only a few years ago. Lack of such normalisation may explain much of the variability in the data in Table 1.

At higher trophic levels in the marine food chain than fish marine birds and mammals do not take up contaminants from their respiratory surfaces as they are air-breathing and concentrations of contaminants in air are extremely low. Thus the only route for contaminant uptake is by their food. So not surprisingly at such trophic levels biomagnification is often observed, in for example sea-birds (Monteiro et al., 1998), in Antarctic food chains (Bargagli, 1998), in seals (Ruus et al., 1999), in whales (Muir et al., 1996) and in polar bears (*Ursus maritimus*) (Atwell et al., 1998). Yet seals and whales are much larger and older than their prey and many seabirds live to be over 30 years old. One might expect that contaminants accumulate with increasing age. The evidence however, is equivocal. Ruus et al. (1999) show that in Harp Seals (*Phoca groenlandicus*) there is an increase in PCB concentration with age, whereas Muir et al. (1996) show only a weak correlation with age in beluga whales (*Delphinapterus leucas*). Female seals excrete contaminants into their milk and lactation is a way of decreasing contaminant load. Seals are able to metabolise PCB congeners with vicinal hydrogen atoms at *meta*- and *para*-positions, whereas there is evidence that cetaceans cannot do so (Metcalfe and Metcalfe, 1997). Boon et al. (1994) have described in detail the structural characteristics of PCBs and divided them into five groups. Groups I and II are highly persistent whereas group III were not persistent in marine birds and mammals. Groups IV and V were persistent in some marine taxa but not in others.

Once inside the organisms the contaminant can be metabolised and may be excreted. In general in small organisms both uptake and elimination occur through the body wall. The importance of gill structures increases with body mass. In fish it is the gills that are the site of passive diffusion of lipophilic chemicals both into and out of the organisms. The absorbed contaminants are distributed among the lipid compartments within the body. This is not the place to discuss uptake and depuration mechanisms, however, Leblanc (1995) and McKay and Fraser (2000) have good discussions on this topic. In general elimination rates for lipophilic contaminants decrease with increasing body mass and with lipophilicity. In a comparative plot Leblanc (1995) showed that daphnids eliminated xenobiotics 10–100 times more rapidly than fish and algae are able to depurate α-HCH, DDT and atrazine significantly more rapidly than daphnids. Thus this inverse relationship between depuration rates of lipophilic contaminants and body mass contributes significantly to explanations of why higher trophic levels (usually larger mass) often have higher concentrations of contaminants.

Yet differences between the structure of organic chemicals is also important. For example, Metcalfe and Metcalfe's (1997) study of PCBs in Lake Ontario, Canada showed that biomagnification occurred with PCB congeners that had octanol–water partition coefficients ($\log K_{ow}$) between 6 and 7.5. Non-*ortho*-congeners were not biomagnified due to their high rates of metabolic clearance.

The concentration of contaminants within organisms thus reflects the balance between the value of the octanol–water partition coefficient, uptake and elimination rates. These factors need to be taken into account when considering data on contaminants and trophic levels. The simplistic assumption, that higher concentrations of contaminant chemicals found at higher trophic levels reflects solely biomagnification processes, is clearly unfounded.

7. Discussion

The conclusions drawn in this paper are far from novel, but do not seem to be appreciated in the general ecological literature on concentrations of xenobiotics in marine food webs. Connell (1990), Leblanc (1995) and McKay and Fraser (2000) have given excellent reviews of this topic and particularly on the mechanisms of

uptake and elimination and should be consulted for further information. So what can ecologists learn from this?

Firstly, the most obvious conclusion from the above analysis is that biomagnification is not a universal rule in marine ecosystems. In particular some contaminants, such as metals, are fairly easily eliminated from organisms and do not accumulate. Only organic Hg of the studied heavy metals shows evidence of non-elimination.

Secondly, the simplest explanation for marine data showing greater concentrations at higher trophic levels (up to fish) is that of passive uptake by diffusion through body surfaces including gills, with elimination rates decreasing with increased body size (Connell, 1990; Leblanc, 1995; McKay and Fraser, 2000).

Thirdly, up to fish there is usually no need to assume that food is the major route for contaminant intake and therefore, that biomagnification is important. However, organisms that have aerial respiration (e.g. sea-birds, reptiles and marine mammals) must take in contaminants via food rather than their body surface and are likely to show biomagnification.

So what should ecologists do to interpret their data? One of the first problems is that often species are collected and yet one knows little or nothing about their trophic status and what level they feed at. Measurement of δ^{15}N in organisms will give a fairly reliable picture of the trophic level at which a species occurs. Many studies, however, show that most species in marine trophic webs are omnivores, (e.g. Atwell et al., 1998). Within trophic levels there is often a great deal of variability both in δ^{15}N values and contaminant levels. Much of this variability will be due to biological variables such as age, stage of gonad development, amount of lipid stored etc and there are also likely to be differences in bioaccumulation rates and elimination rates between species within a trophic level.

A second point from the data analyses is that often methods are not standardised. The commonest failing is using whole body tissues in invertebrates and specific tissues in larger animals. Using similar tissues in different trophic levels may well help to reduce the variability in contaminant data within organisms. In fact there are almost no data available that have examined contaminant concentrations in, say, muscle tissues from invertebrates to mammals. Such data are urgently needed. Thirdly, it has been shown that with organic contaminants normalisation to a concentration based on lipid content is essential. Without such normalisation the results may simply show that in one species (or life stage or at one season) there is a higher lipid content (and hence organic contaminant) than in another species. Leblanc (1995) has shown that phytoplankton, invertebrates and fish have mean lipid contents of 0.5%, 1.8% and 5.4%, respectively. Yet here again there are problems as there is no agreed standardised method for lipid analysis and different extraction methods give different values. Perhaps all that can be expected is consistency within one study and a clear statement of which methods were used. Fourthly, as the mass of an organism increases the ratio of depuration sites to chemical storage capacity decreases. Thus depuration rates for lipophilic substances decreases with organism size. This means that the data also need to be standardised to a unit size in order to properly evaluate whether biomagnification occurs.

Most chemicals enter organisms by means of passive bioconcentration from the soluble phase. Much progress has been made in recent years in the development of predictive models for bioconcentration (McKay and Fraser, 2000). As has been mentioned earlier the octanol–water partition coefficient, (K_{ow}) gives a good descriptor of the likelihood of uptake of a hydrophobic substance. Chemicals with a high K_{ow} have only a small fraction in the dissolved form and thus are not readily taken up by organisms in this phase. However, chemicals with a high K_{ow} adsorb readily to particles and may be taken into the body by this route. For a chemical to biomagnify it must have a bioconcentration factor of $>$114,000. Only a few chemicals have such properties, (i.e. a K_{ow} of $>2 \times 10^6$) such as DDT and PCBs (Leblanc, 1995). Thus again biomagnification is unlikely for most chemicals.

As an ecologist interested in marine sediments one of the major problems is how bioavailable are the contaminants adsorbed to marine sediments. O'Connor and Paul (2000) have recently analysed the US Environmental Protection Authority's EMAP-Estuaries Program data and the NOAA Bioeffects Survey data. They show that no chemical measurement reliably predict sediment toxicity. One of the major problems is that the assumption is that pore-water concentrations are in equilibrium with the sediment (equilibrium partitioning theory, Bierman, 1990). Here the assumption is that data on the bulk sediment concentration and total organic carbon can be used to calculate pore-water concentrations. Yet O'Connor and Paul (2000) show that equilibrium rarely exists and many organic chemicals with high K_{ow} values are insufficiently soluble to show aqueous phase toxicity. Thus we are faced with huge methodological problems to try and resolve what concentrations are actually available to bioconcentrate. As most contaminants adsorb to particles and are deposited in marine sediments this represents a major hurdle to our understanding of bioaccumulation processes (either bioconcentration or biomagnification) in the marine environment.

Whereas in the past simple regressions of bioconcentration (or bioaccumulation) against K_{ow} were used to make predictions, today a more detailed approach using mechanistic models is used. McKay and Fraser (2000) give an excellent summary of the state-of-the-art of such

models. They conclude by stating that to model food webs and biomagnification is a formidable problem especially in relation to model parameterization and laboratory and field validation.

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