CHAPTER 7

Ecological Impacts of Organic Chemicals on Freshwater Ecosystems

Paul K. Sibley[1,*] and Mark L. Hanson[2]

[1]School of Environmental Sciences, University of Guelph, Canada and [2]Department of Environment and Geography, University of Manitoba, Canada

Abstract: The ecological impacts of organic pollutants on freshwater ecosystems have attracted immense scientific, regulatory, and public attention over the past fifty years. In part, this reflects the significant role that freshwater ecosystems play as a repository for anthropogenic chemicals relative to other systems. Some of the most severe ecological impacts have been documented in freshwater ecosystems from persistent organic pollutants (POPs) such as polychlorinated biphenyls, polychlorinated dioxins and furans, and polycyclic aromatic hydrocarbons. Such chemicals can reside for long periods in freshwater sediments, which can then constitute a continual source to the environment even when direct inputs have ceased. Exposure of freshwater biota at lower trophic levels to persistent chemicals can result in transfer to, and ecological impacts at, higher trophic levels through bioaccumulation and biomagnification. In contrast to historically significant organic pollutants, the pervasive nature of new pollutant classes (e.g. pharmaceuticals, polybrominated diphenyl ethers, and perfluorinated surfactants) in global freshwater ecosystems is beginning to be recognized but the full spectrum of their ecological impacts is poorly understood. In this chapter we review documented and potential ecological impacts of organic chemicals in freshwater ecosystems. We focus predominantly on effects at the population, community, and ecosystem levels but, to the extent that our understanding of impacts at these higher levels is predominantly extrapolated from information derived at lower levels, we also include information at the organism and sub-organism level. In addressing each chemical class, impacts on microbial, plant, invertebrate, fish, and fish-eating bird populations are considered where data exists.

INTRODUCTION

The conceptual basis of the ecosystem, *i.e.* a system that includes living organisms interacting with each other and the inorganic components, was defined by Tansley [1]. The hierarchical and interconnected flow of energy and cycling of materials within an ecosystem lead to the concept of trophic structure [2] and subsequent bioenergetic studies of ecosystem development, including relationships between biota (food webs), diversity, and nutrient cycles [3]. From this pioneering work emerged the concept of hierarchical levels of organization within ecosystems (Fig. **1**). This hierarchical framework may be instructive in understanding the ecological impacts of contaminants; that effects at a given level of biological organization can propagate upward, or cascade downward, to other levels [4]; and that effects at one level can be understood mechanistically from information derived at lower levels in the hierarchy and interpreted ecologically from information derived from higher levels [5]. However, the greater the distance between any two levels, the more difficult it is to establish cause-effect relationships and, coupled with the non-linearity of many trophic relationships, clear examples of effects propagating from sub-individual levels to higher levels of organization are rare.

Current regulatory structures for protecting aquatic ecosystems typically rely on extrapolation of data derived from lower levels of biological organization (e.g. whole-organism toxicity tests) as these are often the only data available for criteria-setting. This situation stands in stark contrast to the protection goals of regulatory authorities, and the fundamental premise of ecological risk assessment (ERA), which is the protection of populations and communities. To bridge the gap between regulatory practice and the protection goals of ERA, much effort has been expended on understanding how ecosystems respond to contaminants at different levels of biological organization. Depending on the intensity and duration of exposure to organic contaminants, ecological impacts in the field may include avoidance, extirpation/extinction, loss of diversity and function, and, under severe situations, ecosystem collapse. Such large-scale impacts were observed in Lake Erie in the 1950s and in Great Lakes lake trout and fish-eating bird populations in the 1960s and 1970s [6]. Of course, ecosystems can recover when ameliorative action is taken as was witnessed in Lake Erie in the 1960s when phosphate inputs were reduced. Although examples of population collapse, community disruption, and ecosystem impacts in the field exist, unequivocal cause-effect relationships

***Address correspondence to Paul K. Sibley:** School of Environmental Science, University of Guelph, Ontario, Canada N1G 2W1; Email: psibley@uoguelph.ca

Francisco Sánchez-Bayo, Paul J. van den Brink and Reinier M. Mann (Eds)

between individual contaminants and ecological effects at higher levels of biological organization are difficult to establish due to high biotic/abiotic complexity.

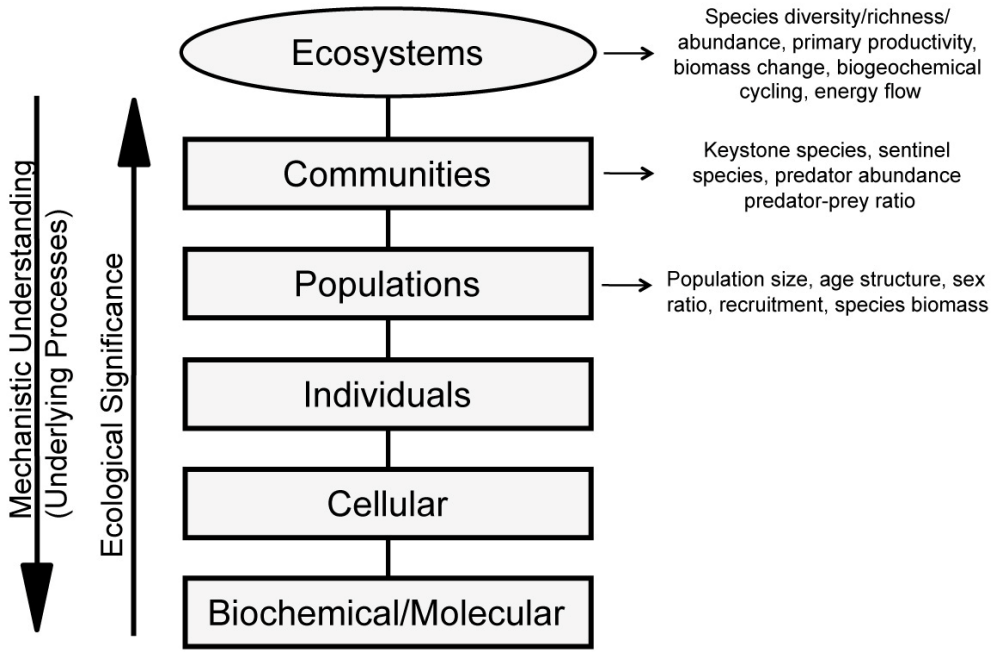

Figure 1: Schematic representation of the hierarchical organization of biological systems

Some understanding of the potential impacts of organic chemicals at higher levels of biological organization can be extrapolated from studies using non-chemical stressors to manipulate whole ecosystems [7, 8]. However, the practical, and arguably ethical, difficulty of manipulating whole ecosystems, communities or populations to understand the ecological impacts of contaminants limits public and scientific acceptance of such approaches. This may be partially overcome by conducting manipulative studies in model aquatic ecosystems (e.g. micro/mesocosms) and considerable knowledge about contaminant impacts at higher levels of biological organization have been derived through the use of such systems. Models can also help to understand potential ecological impacts of contaminants at higher levels of biological organization. Complex ecosystem simulation models have been developed (e.g. Comprehensive Aquatic Systems Model for understanding the impacts of pesticides) but have not been applied extensively because of the large number of explicit/implicit assumptions needed for parameterization, the large amount of data required about the fate and effects of the chemical(s) in an ecosystem, and difficulties related to model validation [4]. Better success has been met with population models and these are commonly applied in ERA [9, 10].

In this chapter, we review the potential ecological impacts of organic contaminants on freshwater ecosystems, focusing on microbes, plants, invertebrates and vertebrates. The chapter is organized by contaminant class, including those such as polychlorinated biphenyls, polychlorinated dioxins/furans, polycyclic aromatic hydrocarbons, and plasticizers (alkyphenol ethoxlylates, bisphenol A) with a long historical presence in the environment and those, such as polybrominated diphenyl ethers, fluorinated surfactants, and pharmaceuticals with a much shorter history. We exclude pesticides as these are covered in Chapter 6 of this book. The scope of this review is largely restricted to population, community, and ecosystem levels of biological organization, but we draw on information from lower levels of biological organization as needed.

HALOGENATED AROMATIC HYDROCARBONS

Halogenated aromatic hydrocarbons (HAHs) are a diverse class of organic chemicals, within which occur some of the most ubiquitous and toxicologically significant chemicals in aquatic ecosystems including: polychlorinated biphenyls (PCBs), polychlorinated dioxins and furans (PCDDs/PCDFs), and polybrominated diphenyl ethers (PBDEs). The unique

physicochemical properties of HAHs including hydrophobicity, low melting points, high octanol-water partition coefficients (Kow), and low volatility reflect the unique properties of halogens, which can comprise a significant percentage of the molecular weight of these compounds. Halogens have high electronegativity, a measure of how strongly atoms attract and hold electrons, and therefore the strength of covalent bonds. Fluorine, chlorine and bromine have electronegativity values of 4.0, 2.0, and 2.8, respectively, which are among the highest in the periodic table. The strong covalent bonds formed by halogens impart high molecular stability and hence a strong propensity to persist in the environment. The environmental persistence of these compounds increases the probability of exposure for environmental receptors while their hydrophobic nature can result in bioaccumulation/bioconcentration and subsequent biomagnification. For many HAHs, the combination of persistence and hydrophobicity has left an indelible imprint on many freshwater ecosystems and yielded a long history of scientific, regulatory, and public scrutiny.

Polychlorinated Biphenyls and Polychlorinated Dioxins/Furans

Background and Chemistry

Polychlorinated biphenyls were introduced in the 1920s as cooling and insulating fluids for industrial transformers and capacitors, fluorescent light ballasts, and as hydraulic fluids in the automotive and related industries [11]. The chemical properties of PCBs that made them ideal for these applications include low flammability and electrical conductance and high thermal and chemical stability. PCBs contain between 1 and 10 chlorine atoms attached in various configurations to biphenyl and were primarily marketed by the Monsanto Corporation between 1930 and 1977 under the trade name Aroclor. There are a total of 209 PCB congeners but only 100 to 150 occurred in formulations that were used and are now ubiquitously dispersed in the global environment [11]. PCBs were first reported in herring gulls and eagles in the mid-1960s [12], and have since been consistently identified in, among other matrices, human and animal adipose tissue, breast milk, and freshwater and marine sediments [13]. Evidence of chronic toxicity in humans and widespread effects in the environment led to implementation of the final PCB ban rule by the EPA in 1979, prohibiting the manufacture, processing, distribution and use of PCBs. PCBs have now been banned for 30 years, but it is estimated that approximately 70% of the PCBs manufactured remain in the environment [14]. In 2001 PCBs were listed as one of the "dirty dozen" POPs under the Stockholm Convention.

In contrast to PCBs, PCDDs/PCDFs have no commercial value and largely occur as historical by-products of the manufacture of organochlorine compounds (e.g. PCBs and pesticides such as 2,4,5-T and pentachlorophenol), the incineration of chlorine-containing substances such as polyvinyl chloride, and chlorine-based bleaching of wood pulp to make paper. Polychlorinated dioxins and furans are also created naturally as pyrolytic by-products of volcanic and forest fire activity. Polychlorinated dioxins and furans contain 1 to 8 chlorine atoms, yielding 75 and 135 possible congeners, respectively. Of these, the 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) congener is the most toxic and PCDD/PCDF congeners having chlorine atoms in the 2,3,7,8 positions appear to be most toxic and bioaccumulative. Once accumulated, clearance of the 2,3,7,8 congeners is extremely low and biomagnification occurs through food chains, at a rate of 3 to 10-fold for each trophic level [15].

Ecotoxicology

The environmental toxicology of PCBs and PCDDs/PCDFs has been well documented [12-14,16-18]. Animal studies indicate that PCBs and PCDDs/PCDFs cause teratogenic, mutagenic, carcinogenic, immunotoxic, and hepatoxic effects and both are known to disrupt endocrine and growth factor systems, including effects on the developing immune, nervous, and reproductive systems [18]. Mechanistically, the toxicity of PCBs and PCDDs/PCDFs is mediated through the aryl hydrocarbon receptor (AhR) which is present in jawed fish, mammals, reptiles, and birds but not primordial fish and invertebrates [19]. For this reason, invertebrates are generally insensitive to PCBs and PCDDs/PCDFs [20] and effects are most commonly observed only in biota of greater evolutionary complexity, although the degree of sensitivity varies considerably among species. Although much effort has been expended to identify cause-effect relationships between specific PCB and PCDD/PCDF congeners and ecological impacts at higher levels of biological organization, this has proven difficult because individual congeners exhibit varying degrees of toxicity and they are taken up and metabolized at different rates as they are passed up food chains [21] leading to different contaminant profiles in aquatic biota over time.

PCBs and PCDDs/PCDFs are hydrophobic. In aquatic ecosystems they may be accumulated through bioconcentration but given their hydrophobic nature and predominant association with sediments and lipids, bioaccumulation through

dietary sources is probably more significant. PCBs and PCDDs/PCDFs are generally metabolized and eliminated slowly from tissues so they not only accumulate but also increase in concentration as they are passed up food chains. Consequently, these chemicals are most commonly found in highest concentrations in predatory species at the top of food chains as documented in the Great Lakes and Arctic regions. In the Great Lakes, PCB concentrations have declined in water, sediments and biota by up to 95% from peak concentrations in the mid-1970s but they remain sufficiently high that they continue to be a major cause of fish consumption advisories [22].

PCBs and PCDDs/PCDFs appear to be relatively non-toxic to freshwater microbial communities at environmental concentrations. Salizzato *et al.* [23] found that the maximum concentration (0.90 □g/g) of PCBs extracted from contaminated sediment, a value well above those typically detected, was below the limit of sensitivity of the Microtox assay. Numerous studies have demonstrated that PCB and PCDD/PCDF congeners can be degraded by microbial communities in contaminated freshwater sediments [24-27]. Aerobic degradation typically involves attack of the carbon ring and subsequent metabolism of the molecule [24] while anaerobic degradation occurs through reductive dechlorination [25]. In general, the rate of dechlorination depends on the relative number and position of chlorine atoms on the molecule and generally decreases with an increase in the number of chlorine substituents. Although rates of metabolism attained in laboratory studies are often high, in situ removal rates are generally exceptionally slow due to poor bioavailability and mass transfer [28]. Nonetheless, it is common to observe population-specific increases in abundances and shifts in microbial community structure in PCB or PCDD/PCDF-contaminated sediments [27].

There is limited information on bioaccumulation and toxicity of PCBs and PCDDs/PCDFs for freshwater plants and algae [29, 30] and evidence of population and community-level effects is rare. Patterson *et al.* [31] evaluated the historical response of diatoms and chlorophytes in sediment cores from a PCB-contaminated freshwater lake. During the period of maximum contamination (estimated peak bioavailable sediment concentrations were 0.5 μg/L), minimal changes were observed in both diatom and chrysophyte assemblages. They hypothesized that the bioavailable fraction of PCBs in lake sediments was too low to cause detrimental effects in the limnetic phytoplankton communities. Kostel *et al.* [32] found that periphyton in a laboratory stream system accumulated up to one order of magnitude greater concentrations than sediment. Periphyton community structure shifted from a diverse diatom-based community to one co-dominated by fewer types of cyanobacteria. Yockim *et al.* [33] estimated bioconcentration factors up to 2083 for 2,3,7,8-TCDD for the freshwater alga *Oedegonium cardiacum* but did not indicate if effects on growth or population size occurred. Residues of up to 7000 ng/g in freshwater macrophytes (*Ceratophyllum* and *Elodea* spp.) were measured in a 30-day mesocosm study on 2,3,7,8-TCDD but no adverse effects were reported [34]. These limited data indicate that significant impacts of PCBs and PCDDs/PCDFs on plants and algae are unlikely at current environmental concentrations but they may serve as an important source of these compounds to higher trophic levels.

Information on the toxicity of PCBs and PCDDs/PCDFs in freshwater invertebrates is also relatively limited. Dillon and Burton [35] found that exposure to some PCB congeners killed 47 to 83% of freshwater fishes and invertebrates after 24 to 48 h at concentrations that were several orders of magnitude higher than those encountered under field conditions. However, most of the PCB congeners tested produced negligible mortality. In the freshwater cnidarian, Hydra aligactis, Adams and Haileselassie [36] estimated LC50s of 5 and 20 mg/L, and found bud regeneration was inhibited at 1 and 4 mg/L, for Aroclors 1016 and 1254, respectively. In these and other studies, effects occurred at concentrations much greater than those measured in freshwater sediments. West *et al.* [37] observed no toxicity in full life cycle exposures with the midge Chironomus tentans and the oligochaete Lumbriculus variegatus up to 9533 ng/g lipid of 2,3,7,8-TCDD. TCDD also had no effect on development and reproduction in the freshwater snail Physa sp. [38, 39], the water flea Daphnia magna [39, 40], the mosquito Aedes aegypti [38], and the aquatic oligocheate Paranais sp. [38]. In contrast, Ashley *et al.* [41] estimated LD50s between 0.03 and 1.5 ng/g body weight for 2,3,7,8-TCDD in a freshwater crayfish and toxicity was characterized by delayed mortality (15 to 40 days after treatment) and reduced activity.

Given the relative insensitivity of invertebrates to PCBs and PCDDs/PCDFs, evidence for population and community-level impacts is rare and often confounded by co-occurrence with other contaminants or non-contaminant factors. For example, De Lange *et al.* [42] found that sediment moderately contaminated by PCBs and PAHs affected the structure but not productivity of benthic macroinvertebrate communities, which they attributed to counteracting effects between contamination and an associated food surplus. Cooper *et al.* [43] compared two urbanized watersheds in Michigan USA, one whose sediments were heavily contaminated with PCBs, PAHs and metals compared to the other. They found fewer

insect taxa, reduced invertebrate index of biotic integrity scores, and higher sediment toxicity in the more industrialized watershed. Although PCBs exceeded the probable effects level at one site, their contribution to the benthic community impacts appeared to be low relative to other contaminants.

Collectively, the weight-of-evidence indicates that significant population or community-level effects from PCBs and PCDDs/PCDFs in invertebrates are improbable at environmentally relevant concentrations. This conclusion is supported by the hazard assessment of Loonen *et al.* [44] who concluded that invertebrates experience reduced hazard relative to fish, fish-eating birds, and mammals. Mechanistically, this has been attributed to the absence of the Ah receptor in invertebrates. However, although effects of PCBs and PCDDs/PCDFs might not be predicted for most invertebrates based on receptor-mediated toxicity, one area that has not been evaluated extensively is potential multi-generational effects resulting from long-term, low-level exposures and this may warrant some consideration in future assessments of these compounds.

Because of their relative insensitivity, association with sediments and occurrence at the base of most food chains, algae, macrophytes and macroinvertebrates play a key role in the bioaccumulation and transfer of PCBs and PCDDs/PCDFs to higher trophic levels. As such, some have been proposed as reliable indicators of PCB contamination in freshwater systems [45] and numerous studies have investigated uptake, metabolism, and trophic transfer by invertebrates of PCBs [46-54] and PCDDs/PCDFs [55-57] in aquatic invertebrates. Zebra mussels (Dreissena polymorpha) accumulated PCB 77 from sediments, diet and water at a rate 10 times more efficient than Lampsilis siloquoidea, the mussel to which they are often attached [58]. Accordingly, high densities of zebra mussels likely influence PCB contaminant dynamics in Great Lakes ecosystems [59]. The freshwater crustacean Mysis relicta played a key role in PCB transfer from sediments into the Lake Champlain food web [60], and Sallenave *et al.* [61] showed that accumulation of PCB 153 in spiked plant material by downstream collectors was enhanced by the presence of both scrapers and shredders in stream mesocosms. Kidd *et al.* [62] estimated that PCBs were accumulated in lower trophic level organisms between 1000 and 100,000 times over surrounding water and sediment concentrations and Rasmussen *et al.* [63] showed that each trophic level contributed a 3.5-fold biomagnification factor for PCBs in Great Lakes lake trout. For TCDDs/TCDFs, Muir *et al.* [55] determined biota-sediment accumulation factors (BSAFs) of 24.6 and 18.6 for crayfish and mussels exposed to TCDF in an experimental lake mesocosm study and BSAFs ranging from 0.31 to 1.62 for uncaged aquatic insects exposed to pulp mill effluent. In laboratory exposures, Loonen *et al.* [56] determined BSAFs of 1.6 and 0.07 for TCDD and octachlorodibenzo-p-dioxin and Pickard and Clarke [57] determined BSAFs ranging from 0.04 to 2.42 for eleven TCDD/TCDF congeners in L. variegatus.

In the Great Lakes region, declines in populations of both fish and fish-eating birds have been causally linked to the presence of PCBs and PCDDs/PCDFs [64, 65] with corresponding though often poorly understood impacts at the community and ecosystem level [66]. Lake trout populations in the lower Great Lakes provide an excellent case study on the role that contaminants likely played in regulating population levels. Lake trout population declines began in the 1930s in response to increased fishing pressure, advancements in fishing technology (e.g. improved fishing line materials), increases in invasive sea lamprey populations, and changes in food web structure caused by invasive fish species [67]. By 1960, a virtual collapse of lake trout populations in the lower Great Lakes had occurred. Extensive re-stocking of fingerlings began in the 1950s and continues to this day but has met with poor success. Hypotheses to explain the slow recovery of Great Lakes lake trout include poor survival to spawning age after stocking and hence insufficient numbers to ensure successful annual recruitment, failure to locate natural spawning grounds due to loss of olfactory acuity in hatchery-reared fish exposed to inappropriate spawning substrates, changes in reproductive performance due to complex changes in prey fish population densities, and exposure to contaminants [67].

There is strong evidence supporting the role of contaminants, particularly PCBs and PCDDs/PCDFs, in lake trout population declines [68]. Initial evidence for the role of contaminants was provided by Mac *et al.* [69] who found that up to 97% of lake trout fry reared in hatcheries between 1978 and 1981 died when exposed to water from the upper Great Lakes. Further, a higher than expected frequency of blue sac disease, which can lead to the death of eggs, was observed in lake trout from Lake Ontario in the 1970s suggesting that maternal transfer of dioxin and dioxin-like compounds to the eggs may have been responsible for the effects. Numerous studies have since established causal relationships between early life stage mortality in lake trout and exposure to PCDDs/PCDFs/PCBs [70]. Although the precise cascade of events from initial exposure to early life stage mortality is not fully understood, as Ah receptor agonists, PCBs and PCDDs/PCDFs are known to affect reproduction and development via disruption of endocrine function [71]. Ankley and Giesy [65], using a weight-of-evidence approach, outlined a series of laboratory and field studies conducted throughout

the 1990s on Lake Ontario lake trout which provide strong evidence to link PCDDs/PCDFs with lake trout population declines. These studies established that early life stage lake trout were exquisitely sensitive to PCDDs/PCDFs and other Ah receptor agonists, that these compounds were transferred maternally from adult fish to eggs, and that effects were consistent with observed pathologies, such as blue sac disease, in field-collected lake trout [70, 72-74]. Comparing the observed effects to residues of PCDDs/PCDFs and PCBs extracted from Lake Ontario sediment cores, Ankley *et al.* [48] concluded that the toxicity predictions are in excellent agreement with available historical data for lake trout population levels and suggest that evidence for recent improvement in natural reproduction is consistent with declining levels of persistent bioaccumulative chemicals in sediments and biota, a conclusion that is supported by Cook *et al.* [70].

With their position atop freshwater food chains, fish-eating birds often represent the most vulnerable group of organisms with respect to the effects of POPs. Population declines of fish-eating birds such as herring gulls, cormorants, and Caspian terns in the Great Lakes have been linked to contaminant-induced reproductive effects [75]. Keith [76] found that reproductive success in herring gulls in Lake Michigan was approximately one third of the normal rate and Gilbertson [77] found that reproductive success was about 10% of expected rates in nesting sites in Lake Ontario. Initially, reproductive failure in fish-eating bird populations was attributed to egg shell thinning resulting from exposure to the pesticide DDT but high rates (up to 30%) of embryo mortality among gull populations [66] and continued observation of developmental and reproductive abnormalities after the ban of DDT in the early 1970s indicated other chemicals were also contributing to population declines. Experiments in which eggs were transferred from clean to contaminated sites for herring gulls [78] and Foster's terns [79] found lower hatching success and behavioral anomalies in adults, thus supporting the contaminant-based theory of observed declines. Many studies documented numerous consistent symptoms including increased embryo and chick mortality, growth retardation, congenital deformities (*i.e.* cross-bill syndrome), feminization of embryos, and abnormal parenting behavior. These effects became known as the Great Lakes Embryo Mortality, Edema, and Deformities Syndrome (GLEMEDS; [80]), a syndrome previously documented in animal studies on known Ah receptor agonists.

Causal evidence that PCBs and PCDDs/PCDFs contributed to population declines in Great Lakes fish-eating birds was developed in the 1990s. Ludwig *et al.* [81] evaluated reproductive success in a population of Caspian terns in Saginaw Bay, Michigan following a one in a hundred-year flood in 1986 that released sediment-bound PCBs. PCBs accumulated rapidly in tern eggs, accounting for 98% of the toxic equivalents (TEQs), and concentrations in the eggs approached the lethal dose required to kill 95% of chicken embryos. The percent of chicks that hatched from first and second clutches the following year was 28% and 0%, respectively, compared to corresponding 5-year rates of 60% and 43% and the 3-year average hatchling deformity rate increased 163 times over the historic rate. Using a weight-of-evidence approach, Ludwig *et al.* [82] reviewed available data from studies on cormorant and Caspian tern populations around the Laurentian Great Lakes to test the hypothesis that deformities in embryos and chicks of these species were caused by contaminants measured as TEQs. Hatching deformities and abnormalities were comparable to those observed in chickens exposed to PCBs and dioxins and were correlated with concentrations of PCBs and TEQs, which were present at concentrations sufficient to cause the effects. Overall, they rejected the null hypothesis and concluded that there was a relationship between the incidence of deformities in both bird species and exposure to planar halogenated compounds measured as TEQs or total PCBs. Giesy *et al.* [83], also using a weight-of-evidence approach, concluded that lethality and deformities in embryos of colonial fish-eating Great Lakes birds were caused by multiple planar dioxin-like compounds which expressed their effects through a common mechanism of action.

With strong evidence for the role of contaminants in causing population declines in Great Lakes fish and fish-eating birds, effects at the community and ecosystem levels might be expected. For example, the primary route of exposure for affected bird populations is contaminated fish, so loss of predatory bird species might be expected to cause changes in fish, and possibly other organism, populations [80] *via* a cascade of trophic interactions. However, understanding trophodynamic changes and shifts in food web structure in the context of contaminants is difficult as non-contaminant drivers of change in ecosystem structure (e.g. introduction of exotic species) must also be considered. Some studies have attempted to address this complexity by examining expected shifts in contaminant profiles in food webs experimentally [49, 84] and *via* modeling (e.g. [51]) but empirical evidence of shifts in community structure and ecosystem function resulting specifically from POPs remains elusive [66].

Numerous studies have measured PCB and PCDD/PCDF residues in adult amphibian and reptile tissues but evidence for effects of PCBs and PCDDs/PCDFs at the population and community level is scant [85, 86]. A study at

a PCB-contaminated site in Peducah, Kentucky found that tissue-borne PCBs were significantly higher in larvae than adults of various anuran species but found no evidence of adverse effects at the population level [87]. PCB tissue levels in frogs in five PCB-contaminated southwestern Michigan wetlands were lower than those in sediments and suggested that the apparent lack of effects on frog populations could be explained by limited contaminant accumulation [88]. Jung and Walker [89] estimated that embryos and tadpoles of green frogs (*Rana clamitans*), leopard frogs (*R. pipiens*), and American toads (*Bufo americanus*) are 100 to 1000-fold less sensitive to TCDD-induced lethality than most fish species. Reeder *et al.* [90] observed a significant shift in sex ratios favoring males, and increased prevalence of intersexuality, in field populations of cricket frogs (*Acris crepitans*) and concluded that the evidence suggested a strong association between population declines and TEQ body burdens. They suggested that amphibian populations could be affected at environmentally relevant concentrations; however, in most studies effects occur at concentrations significantly higher than those typically measured in water and sediments. Based on their work with *X. leavis*, Levine *et al.* [91], suggest that the apparent insensitivity of anurans to dioxins reflects low affinity binding by the Ah receptor. Bishop *et al.* [92] suggested that high mortality in snapping turtles (*Chelydra s. serpentina*) in 1984 and 1985 in Hamilton Harbor, Ontario was strongly correlated with tissue PCB concentrations. Bishop *et al.* [93] observed a significant increase in abnormal development with increasing HAH exposure in snapping turtles eggs at various sites in the lower Great Lakes; the strongest correlations were associated with PCDD/PCDF concentrations. Eisenreich *et al.* [94] found no evidence of immediate effects on embryonic development and hatching success of maternally-exposed snapping turtle eggs collected from the Hudson River, USA relative to those from a reference site; however, high mortality and lower growth rates, correlated with PCB concentrations in the eggs, were observed eight months after hatching.

Polybrominated Diphenyl Ethers

Background and Chemistry

Polybrominated diphenyl ethers (PBDEs) are a diverse group of chemicals that are structurally similar to PCBs, and like PCBs have 209 congeners. Produced as octa-, penta-, or deca-BDE formulations, PBDEs are used primarily as flame retardants in commercial products, including building materials, electronics, furnishings, motor vehicles, plastics, polyurethane foams, and textiles [95, 96]. Commercial PBDE products are typically composed of a complex mixture of congeners, although most mixtures are dominated by one or two specific congeners [97]. The three dominant forms used in industrial manufacturing (deca-, octa- and penta-BDEs) each have specific uses, *i.e.*, penta-BDE (polyurethane foams), octa-BDE (rigid plastics e.g. ABS) deca-BDE (textiles, resins, and rigid plastics). By 1990, worldwide production of PBDEs had surpassed peak production of PCBs; coupled with the fact that PBDEs are not chemically bonded to the materials with which they are associated, and are thus readily released during product use and disposal, they now occur widely in the global environment [98].

PBDEs have low water solubility (typically in the low μg/L range) and log K_{ow} values ranging from 5.9-6.2, 8.4-8.9, and 10 for the penta- octa- and deca-BDE, respectively [99]. The penta-, octa-, and deca-BDEs have thus been detected globally in a wide variety of matrices, both biotic and abiotic [97, 100]. In a recent study of marine water and sediments in Japan, concentrations in water, sediment, and fish and invertebrates were in the low pg/L range, ng/kg range, and ng/g range, respectively [101]. In sediments, deca-BDEs generally dominate the total PBDEs, while in biota tetra-BDE (BDE-47) tends to dominate, an observation consistent with other studies [100]. Not all PBDEs appear to biomagnify and those that do, do not appear to biomagnify to the same degree as PCBs [101, 102]. Further, metabolism, specifically debromination, occurs in a number of species, including fish and microbes [98, 103]. Regional comparisons of total and specific PBDE congener concentrations in the environment indicate that PBDE concentrations in Europe are approximately 10-fold below those in North America [100] and concentrations in arctic marine mammals are currently 10 to 100-fold lower than in temperate species [100, 104]. Currently, the penta-BDE mixture is banned in Europe, the main manufacturer of the penta- and octa-BDEs has begun a phase-out of these congeners, and the deca-BDE, despite industry arguments, is now being banned in jurisdictions across the globe [98]. A series of recent reviews have examined their history, chemistry and toxicology in marine and freshwater ecosystems [95-100, 104, 105].

Ecotoxicology

Due to the chemical nature of PBDEs, and their similarities to PCBs, toxicological research has predominantly focused on freshwater and marine organisms. In mammals, PBDEs may act as hormone mimics affecting thyroid

function, sexual development and behavior; be cytotoxic (increased apoptosis and necrosis); and may be linked to tumor formation and cancer [98]. The mode of action of PBDEs is uncertain; like PCBs and dioxins, PBDEs were originally thought to act *via* the Ah receptor but recent evidence suggests that this may not be the case as the biomarkers associated with AhR activation are not up-regulated following PBDE exposure [97, 99]. PDBEs have been implicated in disruption of thyroid hormone homeostasis in rodents [97, 99], though it is unclear if this mode is shared with other vertebrates and invertebrates, especially those with no analogous hormonal system. Recent work has found evidence for disruption of oxidative phosphorylation and inhibition of complex II of the mitochondrial electron transport chain in fish, especially for hydroxylated forms of PBDEs [106].

The limited toxicological research on PBDEs has generally focused on whole organism or molecular responses and studies investigating potential effects of PBDEs at the population/community level are rare. Indeed, the majority of work has focused on characterizing concentrations of PBDEs in fish, birds, marine mammals and invertebrates [102, 107-111]. When examining microbial populations for their ability to metabolize various PBDEs, a strain of bacteria (*i.e.*, *Burkholderia xenovorans* LB400) was shown to exhibit some toxicity when exposed to mono-BDE. The toxicity was attributed to a metabolite formed during the biotransformation process, and this strain also produced hydroxylated PBDEs which, based on work with vertebrates, are thought to be more toxic than the parent compounds [103, 106]. Several recent reviews [103, 106] reveal the paucity of acute toxicity information for aquatic organisms. However, the limited data that are available indicate that acute toxicity from PBDEs or their metabolites is unlikely. In embryonic zebra fish, the 72-h EC50 for developmental effects (e.g. developmental arrest, edema) was 14.5 μg/L (25 nM) and the adult LC50 after 96 h was between 174 and 232 μg/L (300 and 400 nM) for the hydroxylated BDE-47 [106]. Based on current environmental concentrations and appropriate safety factors, the authors concluded that concerns for wildlife for this PBDE metabolite are unwarranted. The 24-h LC50 to *D. magna* for PBDE congener 153 was >210 μg/L with some chronic effects on reproduction at concentrations >12.5 μg/L [112]. Due to their physicochemical properties (e.g. hydrophobicity), PBDEs are not ideal candidates for microcosm/mesocosm-based toxicity studies and because of their similarities to PCBs (chemical, toxicological, and in terms of co-occurrence) it would be difficult to attribute any observed effects at the population or ecosystem level in the field to PBDEs alone. This issue is highlighted by Jaspers *et al.* [110] who measured a variety of POPs (PCBs, PBDEs, HCHs and DDT) in predatory aquatic and terrestrial bird tissues. Of the seven species examined, six showed no relationship between tissue burden and condition index. The lone exception was the barn owl, which showed negative correlations with PCBs and DDT, which were 100 and 10-fold higher in concentration, respectively, than the PBDEs. The same relative difference between total PCB and PBDE burden in salmonid tissue concentrations have been reported (43,100 ng/g lipid and 2440 ng/g lipid, respectively) in Lake Michigan [102], meaning the attribution of ecological effects solely to PBDEs in the field is unlikely or not occurring at this time.

POLYCYCLIC AROMATIC HYDROCARBONS

Background and Chemistry

Polycyclic aromatic hydrocarbons (PAHs) are a class of POPs comprised of thousands of individual substances that contain two or more fused aromatic rings composed of carbon and hydrogen atoms. PAHs are formed through pyrogenic, petrogenic, diagenetic, and possibly biogenic sources [113]. Pyrogenic sources may be natural, such as forest fires and volcanoes, or industrial, such as the incomplete combustion of fossil fuels, fugitive losses during petroleum extraction, transport, and industrial emissions [114]. Petrogenic sources result from diagenetic processes – low temperature, high-pressure reactions of biogenic materials that occur over geological time scales that lead to the formation of petroleum and other fossil fuels. PAHs are generally hydrophobic and many interact strongly with sedimentary organic carbon [113] and bioaccumulate in aquatic biota, particularly those at lower trophic levels [115, 116]. As such, PAHs are commonly associated with sediments and particulate matter and ecotoxicological concerns have therefore focused on toxicity to aquatic benthic communities and impacts on associated food chains. Historically, only 16 PAHs have been prioritized as environmentally significant and thus received the focus of research; however, it is now recognized that aquatic communities may be exposed to, and potentially affected by, hundreds of PAHs [116] and information on their potential risks are poorly understood.

Ecotoxicology

The toxicology of PAHs in aquatic environments has been well documented and numerous reviews/books are available in relation to bioavailability [113, 115], bioaccumulation [115, 117] and toxicity [114, 118, 119]. PAHs

can adversely affect aquatic organisms physically (e.g. smothering, attenuation of light, habitat modification, and reduced food availability) and directly, *via* toxicity from parent or photosensitized PAHs [114]. The former is most commonly associated with accidental releases of petroleum while the latter results from exposure to PAHs associated with oil or derived from natural or industrial sources. The toxicological effects of PAHs are numerous (see Table **14**.1 in [114]). The primary mode of action of PAHs is narcosis [120]. However, some PAHs act as pro-carcinogens through metabolic formation of DNA adducts, a potentially critical initial step in carcinogenesis [121]. DNA adduct formation has been used as a biomarker of PAH exposure in aquatic organisms [119]. PAHs can also induce immunosuppression [122] as indicated by increased incidences of disease in Japanese medaka exposed to benzo[a]pyrene [123]. PAHs and their derivatives may also affect estrogenic activity. Rainbow trout hepatocytes exposed to anthracene exhibited anti-estrogenic activity, possibly mediated through binding to the Ah receptor [124]. Villeneuve *et al.* [125] found that several PAHs and hydroxylated or methylated PAH derivatives induced estrogenic responses in three separate cell lines.

A unique property of some PAHs is the ability to absorb energy from the ultraviolet spectrum of sunlight, resulting in excited state molecules that, through the subsequent loss of energy, can be several orders of magnitude more toxic than the parent molecules [126-128]. This phenomenon, referred to as phototoxicity, has been demonstrated in freshwater invertebrates [129-132], fish [133-134], and amphibians [135-138]. While most of these studies were conducted under laboratory conditions, PAH-UV interactions in the field have been observed [126, 138, 139] and it has been speculated that synergistic interactions between UV light and PAHs in aquatic habitats may be a contributing factor in amphibian population declines [140]. Others have argued that phototoxicity in the field is ecologically irrelevant because abiotic factors (e.g. dissolved organic carbon), physiological mechanisms (e.g. metabolism/excretion) and physical structures (e.g. integument, burrowing, larval cases) mitigate exposure to UV radiation [141].

Evidence for impacts of PAHs at higher levels of biological organization in the field is scant. Unlike many of the classic POPs, PAHs do not biomagnify [115]. Greatest PAH tissue residues appear to be associated with primary consumers and detritivores in sediments, and tissue concentrations generally decrease with increasing trophic level due to species-specific differences in toxicokinetics and increased biotransformation, especially in vertebrates [115, 142]. Thus, effects on populations and communities are more likely to result from direct exposure to PAH or indirect ecological effects than to food chain transfer and subsequent direct effects at higher trophic levels.

Freshwater microbial communities are both affected by and adaptable to PAHs. In a field-based microcosm study, Baker and Morita [143] found that glucose mineralization and phosphatase levels declined significantly but methane and CO_2 production rates significantly increased in sediment bacterial communities after a 4-week exposure to crude oil designed to mimic a spill. Nitrogen fixation was not affected by 0.1% (v/v) oil, but was reduced after 8 weeks by 1.0% oil. In contrast, Nyman [144] found that exposure of wetland sediment microbial communities to two types of crude oil stimulated bacterial metabolic activity as indicated by measurements of redox potential and respiration. PAHs occur naturally, so it is not surprising that microbial communities have evolved the capacity to degrade them [145], and PAH-degrading capacity is much greater in contaminated soil where selection has favored bacteria capable of withstanding exposure [146]. However, in situations of heavy contamination (e.g. oil spill), ecosystem integrity and function may be affected due to lower microbial diversity as this reduction disrupts the tight coupling and interdependence among consortia, and between consortia and grazers. For example, Nyman [144] observed an increase in metabolic activity and oil degrading activity in their wetland sediment study, but this came at the expense of microbial diversity, with tolerant species becoming dominant as sensitive species declined in abundance.

The toxicity of PAHs to freshwater algae and macrophytes has been evaluated in laboratory and field studies. Bott and Rogenmuser [147] exposed algal communities to three oil extracts in stream microcosms for several weeks. No. 2 fuel oil extracts depressed algal biomass (measured as chlorophyll *a*), decreased diatom occurrence, and resulted in dominance by blue-green algae. Used crankcase oil extracts also depressed biomass, but Nigerian crude extracts did not, and both of these extracts had less effect on algal community composition than did the No. 2 extracts. Marwood *et al.* [148] observed effects of PAHs at environmentally relevant concentrations on photosynthesis in natural algal assemblages and attributed this to phototoxicity. Burk *et al.* [149] found that total plant cover, total and mean number of species, and Shannon diversity declined progressively for two years after an accidental oil spill in a marsh and eighteen species found before the spill were absent the following season. However, the vegetation of the marsh showed substantial recovery by the third and fourth years. McGlynn and Livingston [150] modeled

adsorption/desorption and potential effects of sediment PAHs at low concentrations by rooted aquatic plants in field and laboratory experiments. The macrophytes' roots assimilated PAHs and the assimilation exhibited saturation. Growth of the macrophytes was inhibited by PAHs but at concentrations several orders of magnitude greater than threshold effects levels for aquatic animals.

Bestari *et al.* [151] and Sibley *et al.* [152, 153] exposed freshwater plankton communities to creosote in microcosms for 83 days at concentrations ranging from 0.06 to 109 mg/L. Creosote had no direct toxic effect on phytoplankton whose population densities and diversity in all treatments exceeded those in the controls and exhibited a parabolic relationship relative to both time and total PAH [152]. In contrast, zooplankton abundance and diversity was significantly reduced by creosote, with a 7-day community-level no-effect concentration of 5.6 μg/L [153]. The zooplankton community was dominated by rotifers, which proliferated at the expense of more sensitive cladocerans and copepods. Recovery to pre-treatment abundance levels occurred in all concentrations by the end of the 83-day exposure. The growth of phytoplankton populations appeared to be stimulated by both indirect (lower grazing pressure from zooplankton) and direct (hormetic stimulation by PAHs) effects.

Several studies have examined the response of freshwater benthic macroinvertebrate communities to PAHs. Crunkilton *et al.* [154] monitored the response of benthic macro-invertebrates in a small Missouri, USA stream into which 1.5 million liters of domestic crude oil had been spilled. Sensitive members of the benthic community (aquatic insects, mussels, snails) declined to <0.1% of expected abundance 25 days after the spill and species diversity indices and the abundance of mayfly and stonefly genera were below water quality criteria for Missouri streams up to 11 months after the spill. The impacts were attributed to physical obstruction of both substrate and organisms, and PAH toxicity. West *et al.* [155] evaluated the effectiveness of a carbonaceous resin to reduce the bioavailability of PAHs in field-contaminated sediments as a basis for potential remediation using laboratory toxicity tests and field colonization studies. The resin significantly reduced pore water concentrations of eight measured PAHs in both laboratory and field sediments. In laboratory tests, bioaccumulation and phototoxicity in L. variegatus were significantly reduced; in the field-deployed sediments, the resin amendment also decreased pore water PAH concentrations but did not improve benthic invertebrate colonization. Den Besten *et al.* [115] investigated impacts of PAH-contaminated sediments on benthic macroinvertebrates in the Rhine-Muese Delta in The Netherlands. Highly contaminated sediments contained significantly fewer taxa, had lower species diversity compared to reference sites, and produced significant toxicity in sediment bioassays with the invertebrates C. riparius and D. magna. De Lange *et al.* [156] evaluated seasonal variation and bioavailability of PAH in contaminated floodplain lake sediments in relation to benthic invertebrate community structure. While sediment-associated PAH concentrations occurred at levels at which effects were predicted, biovailability was low and the PAHs were not associated with observed impacts on benthic community structure.

Cooper *et al.* [43] compared benthic community and fish population structure in two sub-watershed wetlands of a western Michigan lake, one of which is highly contaminated with PAHs and metals as a result of a long history of industrial activity. Significantly fewer insect taxa, reduced fish species richness and catch per unit effort, and lower invertebrate and fish index of biotic integrity scores were found in the industrialized watershed. Cormier *et al.* [157] used a formal strength-of-evidence methodology [158] to infer causes of impairments at two sites in the Little Scioto River, Ohio, USA, which is heavily contaminated by sediment PAH. At the upstream site, they concluded that impairment of the benthic community and fish populations was due to altered habitat substrate (predominance of fine-textured sediment) and low dissolved oxygen. At the downstream site, impacts included lower diversity and dominance by pollution-tolerant invertebrates and reduced fish growth, elevated PAH tissue concentrations and increased incidences of abnormalities in fish, all of which could be causatively explained by concentrations of sediment PAHs. Lesko *et al.* [159] assessed the effects of contaminated sediments on reproductive potential of female brown bullhead (*Ameiurus nebulosus*) collected from the Black and Cuyahoga Rivers, Ohio, both contaminated with metals, PAHs and PCBs. Females from the most contaminated (Cuyahoga) river had higher fecundity and the population size was larger compared to the reference river, which they attributed to an enhanced food supply due to reduced competition from predators. However, fish diversity in the Cuyohoga River was lower and incidences of tumors higher relative to the reference river [160]. Evidence that PAHs can act as endocrine disrupters has largely been developed for fish [71, 161, 162]. While studies to date have not linked endocrine effects directly to population or community-level effects, evidence that PAHs may impair reproduction in fish, either through altered sex steroid metabolism or biosynthesis (see [71] for examples), reduced growth or abnormalities in larval fish [163] suggest that endocrine-induced population effects are possible.

Studies investigating the effects of PAHs on amphibians and reptiles have largely focused on organism and physiological responses, either through direct exposure to PAHs [136, 164, 165] or synergistic exposure to UV light as described above. Few studies have examined the effects of PAHs at the population and higher levels of biological organization in amphibians. Physiologically, amphibian responses to PAHs are similar to other vertebrates [118]. Lefcort *et al.* [166] studied the effects of oil and silt on the growth and metamorphosis of larval mole salamanders, *Ambystoma opacum* and *A. tigrinum tigrinum* in oil-contaminated ponds and outdoor microcosms treated with used motor oil. In both test systems, both species had reduced size and weight compared to controls that was attributed to an indirect effect of reduced algal growth (salamander food) and direct toxic effects.

FLUORINATED SURFACTANTS

Background and Chemistry

Surfactants are surface-active materials that, at low concentrations, are capable of reducing the surface tension of a liquid *via* selective adsorption at the interface [167]. Surfactant molecules are amphiphilic, characterized by a hydrophilic (water-soluble) 'head' group attached to a hydrophobic (water-insoluble) 'tail' portion. In conventional, hydrocarbon-based surfactants, the hydrophobe is typically an oleophilic (lipid soluble) hydrocarbon. In perfluorinated surfactants (PFSs) fluorine atoms replace hydrogen atoms on the hydrophobe. The replacement of hydrogen with highly electronegative fluorine atoms on the hydrophobe renders PFSs both hydrophobic and oleophobic, capable of repelling both water and oils. Increasing the number of fluorine atoms in the hydrophobe increases chemical stability as bond strength generally increases with an increase in the number of fluorine constituents [168]. The exceptional persistence rendered by the high molecular stability has led to the detection of PFSs in a variety of biotic and abiotic matrices on a global scale [169-172].

The global pervasiveness of PFSs reflects both a long history of manufacture (since the 1950s) and widespread use as surface treatments for carpets, fabrics, and paper products to repel soil, oil, and water and applications such as fire fighting foams, adhesives, electronic insulators, cosmetics, cleaners, among others [167, 170]. Recently, concerns over the occurrence of perfluorooctane sulfonic acid (PFOS) in the environment, especially in sensitive Arctic regions, resulted in a cessation of production in 2000 by 3M Corporation and the recent inclusion of PFOS under Annex B of the Stockholm Convention on POPs, indicating that its use should be restricted.

Ecotoxicology

Environmental concerns about PFSs have predominantly focused on two compounds: perfluorooctanoic acid (PFOA) and PFOS. Over the past decade, the toxicity of PFOS and PFOA to environmental receptors has been well studied as reviewed in [170, 171, 172]. However, with the exception of the microcosm studies described below, most of this work has been conducted at the organism level.

Several studies have assessed the toxicity of PFSs in freshwater macrophytes and algae. Boudreau *et al.* [174] and Boudreau [175] assessed the toxicity of PFOS and PFOA in the algae *Chlorella vulgaris* and *Pseudokirchneriella subcapitata*, and the aquatic plant *Lemna gibba*, under laboratory conditions at chain lengths of 4 to 7 carbons. In tests with PFOS, 96-h growth inhibition NOEC values were 5.3 and 8.2 mg/L for *P. subcapitata* and *C. vulgaris*, respectively, and 6.6 mg/L for *L. gibba* (wet weight). In tests with PFOA, laboratory EC10 values for growth ranged from 5.7 to 59.4 mg/L for *C. vulgaris* (96-h) and *L. gibba* (7 d), respectively [175]. Colombo *et al.* [176] calculated a NOEC value of 12.5 mg/L for growth inhibition in *P. subcapitata* exposed to the ammonium perfluorooctanoate. Liu *et al.* [177] assessed four perfluorocarboxylates and two sulfonates to the alga *Scenedesmus obliquus* and found that toxicity based on cell density ranged from none (PFOA) to 21.6 mg/L (perfluorotetradecanoic acid). Latal *et al.* [178] showed that perfluoro-hexanoic, -heptanoic, -octonoic, and -nonanoic acid were more toxic than PFOS and PFOA to three species of algae (LC50s range: 6.0-24.3 mg/L). Blue-green and diatom species were comparable in sensitivity but both were more sensitive than green algal species. In an outdoor microcosm study, Boudreau *et al.* [174] determined a 42-day NOEC (frond number) for PFOS of 0.2 mg/L for a population of *L. gibba*. Hanson *et al.* [179, 180] estimated NOEC values in excess of 0.3 mg/L and 23.9 mg/L for PFOS and PFOA, respectively, for two species of *Myriophyllum* in outdoor microcosm studies.

With one exception, freshwater invertebrates appear to be relatively insensitive to PFSs. Boudreau *et al.* [174] estimated NOEC values for immobility in 48-h exposures of 0.8 and 13.6 mg/L for *D. magna* and *D. pulicaria*.

NOEC values in tests with PFOA for both *Daphnia* species indicated reduced toxicity relative to PFOS [175]. In both cases, daphnids were only sensitive to carbon chain lengths ≥ 8. Ji *et al.* [181] estimated LC50s of 17.95 mg/L for PFOS and 199.51 mg/L for PFOA for the daphnid *Moina macrocopa*, which is approximately twice the LC50 determined for *D. magna*. In a 7-day chronic test, *M. macrocopa* experienced significantly reduced reproduction at 0.31 mg/L for PFOS, which was approximately seven times lower than the effect concentrations observed over the 21-day exposure in *D. magna*. The greatest toxicity observed for PFOS in any aquatic species is the 20-day LC50 of 9.2 μg/L reported for *C. tentans* [182]. In the same test, *C. tentans* did not respond to PFOA in 10-day exposures at concentrations up to 100 mg/L. In a series of indoor microcosm studies, PFOS caused a significant reduction in zooplankton abundance and altered community structure at concentrations ≥10 mg/L [183]. In a similar test with PFOA, Sanderson *et al.* [184] determined a lowest observed effect concentration (LOEC) of between 10 and 70 mg/L depending on taxonomic group. In these studies, zooplankton communities became dominated by rotifers with simultaneous declines in cladoceran and copepod species at the highest concentrations. In a 35-day outdoor microcosm study, Boudreau *et al.* [174] estimated a community-level NOEC of 3.0 mg/L for zooplankton, with significant declines in zooplankton abundance at 30 mg/L. Kannan *et al.* [185] estimated a bioconcentration factor of approximately 1000 for PFOS in Great Lakes benthic invertebrates and Higgins *et al.* [186] estimated lipid-normalized BSAF values of 33 and 42 for PFOA and PFOS indicating that both compounds may be accumulated from sediments and thus available for trophic transfer in freshwater food chains.

In fish, studies indicate that toxicity thresholds of PFSs are typically much higher than environmental concentrations. Du *et al.* [187] observed no mortality in zebra fish exposed to PFOS in a 70-day exposure but did observe significant declines in growth and various biochemical and genetic endpoints at concentrations as low as 50 μg/L. Hagenaars *et al.* [188] also found no mortality in carp fry exposed to PFOS concentrations up to 1 mg/L but did observe significantly reduced condition factor and liver indices as low 0.1 mg/L. Colombo *et al.* [176] estimated a 96-h LC50 for the ammonium salt of PFOA of 400 mg/L. PFOS and PFOA are hepatotoxic, affecting hepatocyte membranes indicative of necrosis and interfere with fatty acid metabolism [189]. PFOS and PFOA exposures can also decrease circulating sex steroids in fish depending on species, age, and sex [189, 190]. Contrary to mammalian studies, PFOS and PFOA appear to be relatively weak peroxisome proliferators in fish [189]. In freshwater fish, PFOS and PFOA bind tightly to serum proteins [191] and bioaccumulate (from highest to lowest) in fish in the blood, kidney, liver, and gall bladder [192]. Kannan *et al.* [185] found that PFOS concentrations in Chinook salmon were 20 times greater than in their prey species and Furdui *et al.* [193] estimated log bioconcentration factors of 4.1 and 3.8 for PFOS and PFOA, respectively, in Great Lakes lake trout. These data provide evidence of food chain transfer of PFAs. Interestingly, Kannan *et al.* [195] found notable concentrations of PFOS in fish eggs suggesting oviparous transfer.

Numerous studies have measured residues of PFSs in aquatic fish-eating birds [169, 185] but few have assessed toxicity. Newsted *et al.* [194] determined a 5-day LD50 of 150 mg/kg body weight in young (2-day old) mallard ducks exposed to food-borne PFOS. The concentration of PFOS in mallard livers associated with mortality was at least 50-fold greater than the single maximum concentration that has been measured in livers of avian wildlife indicating low risk. Adult mallards fed PFOS up to 150 mg/kg feed showed no treatment-related effects [195]. Based on this work, they estimated an avian toxicity reference value of 0.021 mg/kg body weight per day. Kannan *et al.* [185] recorded the highest concentrations of PFOS from bald eagles in the Great Lakes and estimated a biomagnification factor of 10 to 20. Excretion of PFOS in bald eagles appears to be more rapid than classical POPs [185], but the potential for binding of PFOS with serum proteins and production of metabolites whose toxicity is poorly understood, warrants further investigation with respect to potential risks to fish-eating bird populations.

The occurrence and toxicity of PFSs in amphibians and reptiles is limited to only a few studies. PFS concentrations ranging from 137 to 250 ng/g (wet weight) have been measured in green frogs, yellow-blotched map turtles, and snapping turtles from the Great Lakes [185]. Ankley *et al.* [196] observed reduced growth and delayed metamorphosis, which can impact population stability, in northern leopard frogs at 3 mg/L PFOS and hypothesized that this may have been the result of impaired thyroid function.

The weight of evidence indicates that PFSs pose limited risks to freshwater organisms as toxicity thresholds are typically well above concentrations of PFAs measured in the field. Beach *et al.* [171] derived protective screening-level concentrations for PFOS of 2.3 mg/L for freshwater plants and algae and 1.2 μg/L for aquatic invertebrates, the latter value reflecting the sensitivity of *C. tentans* [182]. A tissue-based threshold value of 87 mg/kg wet weight was

determined to be protective of fish. Collectively, the evidence does not support a causal link between current PFS contamination and population or community-level impacts in aquatic systems.

PHARMACEUTICALS AND PERSONAL CARE PRODUCTS

Background and Chemistry

Pharmaceuticals and personal care products (PPCPs) encompass a wide variety of chemicals and applications, including the cure and prevention of disease in humans and livestock, diagnostic treatment (e.g. x-ray contrast media), growth promotion in livestock, and chemical additives (e.g. musk fragrances, antibacterial agents) in personal care products [197]. Thousands of PPCPs are produced and used daily around the world, in quantities that now approach those typical of agrochemicals [198]. Sources of PPCPs to freshwater environments include wastewater treatment effluents, run-off from agricultural fields amended with manure and sewage, direct addition *via* livestock excretion, and leaching from landfill sites.

In contrast to many of the legacy chemicals addressed in this chapter, knowledge of pharmaceuticals in the environment, and the attendant concerns for human and environmental health, emerged only in the late 1990s. While evidence of hormonally active pharmaceuticals date back to the 1960s [199], the pervasive nature of PPCPs has only recently been brought to light because of advancements in analytical technology that made it possible to detect PPCPs at the low concentrations at which they typically occur. In North America, the widespread occurrence of PPCPs in surface waters was documented by Kolpin *et al.* [200] who identified 82 compounds from surface waters of 139 streams with many compounds co-occurring. Many studies have since added to the list of PPCPs known to be present in the environment [201]. These studies show that PPCPs occur predominantly at sub-µg/L concentrations. However, although the majority of PPCPs occur at low concentrations and degrade rapidly under most environmental conditions, continual addition to the environment renders them effectively "pseudo-persistent" [202]. Moreover, pharmaceuticals are designed to elicit biological effects, which is the basis of their therapeutic activity [203]. The combination of pseudo-persistence and biological activity has lead to uncertainty about how PPCPs will behave toxicologically in the environment and legitimate questions about potential risks to environmental receptors.

Ecotoxicology

The fate and effects of PPCPs in aquatic systems has been summarized in several reviews [203-209]. There is general consensus that acute exposure of aquatic organisms to PPCPs carries negligible risk [210, 211]. In a review of over 360 acute toxicity endpoints in freshwater organisms for 107 human PPCPs, Webb [212] found that <10% were toxic at concentrations <1 mg/L, a value approximately 3 to 4 orders of magnitude above concentrations typically measured in freshwater ecosystems. Thus, interest in PPCPs from an ecological impacts perspective is presently focused on potential impacts from chronic exposures. Here, ecological impacts, if any, will likely depend on the type of PPCP. For example, Fent *et al.* [210] found that chronic LOECs in laboratory test species are about two orders of magnitude greater than maximum concentrations in sewage treatment plant effluents. However, their assessment did not include antibiotics or hormones, both of which warrant additional detailed investigation regarding potential risks to aquatic biota [205, 206].

One of the best studied PPCPs is ethynylestradiol (EE2), a synthetic compound widely used in birth control formulations and commonly detected in sewage treatment plant effluents and biosolids. As a hormone mimic, EE2 is a potent endocrine disrupting agent in freshwater vertebrates and has been implicated in a number of cases of sexual disruptions reported in freshwater fish exposed to sewage treatment effluent [213-214]. EE2 induces synthesis of the egg yolk precursor vitellogenin in male and juvenile fish, can cause increased incidences of intersex (gonads possess features of both sexes), feminization of male fish and reduced fertilization success [215, 216]. However, few studies have linked these changes to actual changes at higher levels of biological organization. One exception is the study of Kidd *et al.* [217] who showed that chronic exposure to environmentally relevant concentrations of EE2 over several years led to the collapse of a population of fathead minnows in a whole lake exposure. In that study, Palace *et al.* [216] found evidence of intersex and inhibited development of testicular tissue in males of pearl dace (*Margariscus margarita*) and suggested a trend toward reduced population abundance and smaller young-of-the-year size classes in the EE2-treated lake. Interestingly, and reflective of the transient nature of PPCP contamination, fish populations were observed to recover after exposure was halted. Watts *et al.* [218] and Dussault *et al.* [219] found no evidence for effects of EE2 in *C. riparius* and *C. tentans*, respectively, in life cycle tests. However, Dussault *et al.* [220]

showed that *C. tentans* and *H. azteca* accumulated EE2 (bioaccumulation factors of 31 and 142, respectively) and could therefore serve as a source of this compound to vertebrate receptors at higher trophic levels. Jensen *et al.* [221] estimated an EC50 of 0.011 µg/L for 17-α-trenbolone, an endocrine-active growth additive used in livestock, in fathead minnows; this concentration is comparable to those measured in beef cattle feedlot runoff [222].

Evidence for effects of non-endocrine PPCPs at higher levels of biological organization is rare but may occur in microbial populations exposed to antibiotics and veterinary medicines. In a recent review of the occurrence and toxicity of antibiotics in aquatic ecosystems, Kümmerer [207] suggests that bacteria and microalgae are 2 to 3 orders of magnitude more sensitive than organisms at higher trophic levels so the prospect for effects on microbial communities cannot be discounted. Indeed, Backhaus and Grimme [223], in a bioluminescence inhibition test with *Vibrio fischeri*, found toxic effect values (EC10) for two antibiotics in the range of concentrations expected in surface waters. Tetracycline was found to disrupt nitrification at concentrations found in some freshwater sediments [224]. Of particular concern with this class of pharmaceuticals is the potential for the development of resistance in freshwater bacteria and this has been demonstrated in natural bacterial populations in sediments associated with aquaculture [207, 225, 226].

Richards *et al.* [227] evaluated the effects of a PPCP mixture composed of ibuprofen, fluoxetine, and ciprofloxacin at individual concentrations of 10, 100, and 1000 µg/L on populations of macrophytes (*L. gibba* and *Myriophyllum sibiricum*), plankton, and bacterioplankton in a 35-day microcosm study. Significant decreases in growth of the plant species at intermediate and high concentrations and eventual loss of plant populations in the high treatment were observed. A significant increase in overall abundance and a significant decrease in diversity of phytoplankton occurred at the high concentration. The higher abundance reflected a large increase in one species that dominated the phytoplankton community; other phytoplankton species were unaffected or were eliminated, explaining the lower diversity. Similarly, zooplankton increased in abundance and had reduced diversity at the highest concentration. The mixture had no effect on bacterioplankton abundance. The authors concluded that the individual risks posed by these compounds in freshwater ecosystems were negligible.

In a 49-day microcosm study (34 days exposure and 14 days recovery), to a four-tetracycline mixture at individual concentrations of 10, 30, 100, and 300 µg/L, Wilson *et al.* [228] measured biomass production, community respiration, and primary productivity, as well as phytoplankton and zooplankton community responses. Phytoplankton abundance and community respiration decreased significantly at the two largest concentrations but primary productivity was unaffected. Community metabolism (ratio of productivity to respiration) decreased significantly at the two greatest concentrations due to significant increases in respiration. Zooplankton were not affected by the tetracycline mixture. The effects observed in this study are approximately 2 orders of magnitude greater than concentrations expected for tetracyclines in freshwater systems. Hillis *et al.* [229] evaluated the effect of monensin, an antibiotic commonly used in beef and poultry, on zooplankton communities in a 50-day microcosm study at concentrations ranging from 0.5 to 500 µg/L. The community-level NOEC (50 µg/L) was approximately 50 times greater than environmental concentrations. McGregor *et al.* [230] observed no impacts of monensin on macrophyte populations up to 100 µg/L in a microcosm study. Sanderson *et al.* [231] evaluated ivermectin, a commonly applied anti-helmenthic drug that has been shown to be highly toxic to aquatic invertebrates, in a long-term (250 day) microcosm study. They demonstrated that ivermectin could pose risks to aquatic organisms at or below the predicted environmental concentrations.

Overall, while there are some exceptions for classes of PPCPs such as estrogens and antibiotics, the weight of evidence indicates that the probability of acute or chronic effects at higher levels of biological organization in aquatic ecosystems is small. Indeed, based on a simple hazard approach comparing the ratio of the predicted effects concentration (PEC) and the predicted no effects concentration (PNEC), Tarazona *et al.* [232] suggest that the likelihood of observing ecosystem-level effects would be expected at ratios of approximately 10 or higher. Such high PEC/PNEC ratios for PPCPs are rarely observed in freshwater environments.

COMPOUNDS IN PLASTICS

Background and Chemistry

The production of plastic yields a variety of potential environmental contaminants. The two most common, which are addressed here, are bisphenol A (BPA, 4,4'-dihydroxy-2,2-diphenylpropane) and nonylphenol (NP), a member of

the alkylphenol ethoxylates (APEs). BPA is a key building block used to produce polycarbonate plastics and epoxy resins [233]. Polycarbonates are incorporated into sheeting, glazing, bottles and storage containers and epoxy resins are used as protective coatings on buildings, boats and vehicles; collectively, this usage accounts for 95% of BPA in the plastics industry [234]. NP is the basis for non-ionic surfactants commonly used in the manufacture of industrial and domestic detergents, pesticide formulations, emulsifier and dispersing formulations, cosmetics, and paints.

BPA is not especially stable in the end product and has been observed to migrate into surrounding environmental matrices. Although BPA is relatively short-lived in the environment (the half-life in water ranges from several hours to a few days depending on initial conditions [235]), continuous inputs and the ubiquity of plastics in the environment, has resulted in routine detection of this compound [233, 236]. BPA is most commonly detected in waters downstream of wastewater treatment plants with concentrations in the effluent typically an order of magnitude greater than in receiving waters [235, 237]. In a review of BPA concentrations in rivers and groundwater, Sharma *et al.* [235] found BPA occurred predominantly at low ng/L and low μg/L range, respectively. In a recent exposure analysis for North America and Europe, median BPA concentrations in freshwater systems were 0.081 and 0.01 μg/L and 0.6 and 3.4 ng/g in sediments, respectively, [238]. Despite low persistence and generally reduced global presence relative to other organic contaminants, BPA has attracted attention from an ecotoxicological perspective because it can bioaccumulate and has been shown to act as an estrogen mimic. For example, BPA is structurally similar to the potent estrogen diethylstilbestrol and has been shown in the yeast estrogen assay to bind and activate the estrogen receptor in vertebrates [233, 239, 240].

NP enters the environment *via* industrial and commercial sources [241, 242]. Due to their occurrence in cleaning agents, high concentrations of NP ethoxylates enter wastewater treatment plants. Here, metabolic degradation leads to the production of NP, which is released into receiving waters [241]. Not surprisingly, NP has been widely detected in systems with wastewater inputs, with the resulting distribution between environmental compartments driven primarily by its physicochemical properties. Due to low water solubility and a K_{ow} >4, fugacity modeling has shown that NP partitions preferentially into sediment, with concentrations downstream of inputs reaching the mg/Kg range compared to low μg/L for water [241, 243]. NP undergoes significant degradation in the water column, with a half-life of a few days [244], but in sediments, half-lives >60 years have been reported [241]. The bioaccumulation potential of NP is generally low to moderate [242], although recent work with zebra fish estimated BCFs >1000 [243]. As an endocrine disruptor, NP can impair reproduction and sexual development as has been shown in fish [241, 245-247]. These estrogenic effects are more pronounced in NP relative to the parent ethoxylate forms and current environmental concentrations may result in population-level effects *via* effects on reproductive fitness [236, 246].

Ecotoxicology

There is a reasonable body of literature examining higher-level effects of NP in aquatic ecosystems. The most extensive is a series of papers that describe the impacts of NP on sediment dwelling nematode, plankton and microbial communities in 230 L aquatic microcosms over an 8-week application phase and a 6-week dissipation phase. The nematode community initially had high abundance and low Shannon diversity, with dominance by one species, *Eumonhystera filiformis* [248]. At week 7, abundances declined and diversity increased but did not correspond to the NP concentrations. The maturity index was the only response that showed a treatment-related response; being significantly lower at the highest concentration (3.4 mg/kg sediment) relative to controls and other treatments. The relative insensitivity of nematodes was attributed to decreased bioavailability due to binding of the NP to the cationic groups in the sediment as a result of the pH of this particular test system. Changes in phytoplankton and periphyton species richness and diversity were not correlated with NP concentrations as the measured NP concentrations were approximately 10-fold lower than those known to cause direct toxicity [249]. However, changes in community composition of phytoplankton were noted, with Conjugatophyceae, which were dominant in all microcosms during the pre-treatment period, being dominant only in the controls and lowest NP concentration post-treatment. In contrast, Cyanophyceae came to dominate at intermediate and higher test concentrations. The authors interpreted this trend as evidence for differential grazing by zooplankton, specifically decreased grazing pressure at higher NP exposures due to direct toxicity on the zooplankton through estrogenic effects [250]. Abundances of copepod larvae were the most severely affected, with declines up to 95% at 200 mg/L NP, with no recovery observed during the 6-week post-treatment period in the three greatest concentrations. Cladocerans were less sensitive, recovering in all but the highest NP exposure, a response that may have been

facilitated by the shift in phytoplankton to smaller species [250]. Time-dependent NOEC values for the community ranged from 19 to 44 mg/L. Interestingly, these NOEC values are lower than those of many single species laboratory invertebrate tests [242], though comparable to that estimated for *D. magna* (EC50 of 16.5 mg/L for population growth (*r*) in a 21-day test) [251]. In a subsequent study, using the same microcosms, Hense *et al.* [252] concluded that decreases in zooplankton reproduction, attributed to the endocrine effects of NP, resulted in a delayed shift in phytoplankton community structure and increases in rotifera, due to reduced grazing and competitive pressures. Microbial communities, specifically bacteria and microfungi in sediments, tended to increase in abundance with elevated NP concentrations in these test systems, with only a slight change in the overall microbial community [253]. This could be attributed to increased food resources due to zooplankton mortality at higher NP exposures, as has been observed elsewhere with mass zooplankton and benthos mortality [254]. In microbial microcosm test systems lacking sediment, zooplankton and benthos, water borne microbial communities showed no significant differences in diversity up to 5 mg/L NP [244].

In fish, NP has been shown to affect behavior and survival and, through estrogenic effects, reproduction [245]. Indeed, field populations of freshwater fish exposed to NP *via* wastewater effluents consistently show endocrine modulated effects such as vitellogenin expression, gonadal abnormalities, reductions in circulating testosterone and reproductive dysfunction, and reductions in the gonadosomatic index, all of which can result in population-level effects through impaired fecundity [241]. However, while some studies show a correlation between NP and these effects downstream of wastewater effluents [255], assigning direct causality to NP when many other contaminants that share a mode of action and input source co-occur is difficult. A modeling exercise using data from laboratory and field studies was conducted to examine the potential impacts on populations of brook trout (*Salvelinus fontinalis*) and fathead minnows exposed to NP for three years at 1 and 30 µg/L [245]. Depending on model parameterization, they predicted an increase in population size of 17% or a decline by 28% at 30 µg/L but no significant change at 1 µg/L NP. Fathead minnows showed a similar response, with population reductions up to 21% and a shortened spawning season at 30 µg/L, but full recovery was anticipated within two years after exposure.

There is a robust body of knowledge on the acute and chronic effects of BPA to a suite of aquatic organisms at the individual level under laboratory conditions (see review by Mihaich *et al.* [256]). In acute exposures, EC50 and LC50 values (24 to 96 h) are typically in the mg/L range for invertebrates (1.1 to 16 mg/L), while chronic testing for invertebrates found NOEC values ranging from 0.25 mg/L in the snail *Marisa cornuarietis* for female growth to >3 mg/L for *D. magna* reproduction [256]. Primary producers appear to be slightly less sensitive than invertebrates. For example, the EC50 for growth for the diatom *Skeletonema costatum* was 2.5 mg/L, while the EC50 for growth for *L. gibba* was 32 mg/L [256]. Based on currently measured environmental concentrations, BPA is unlikely to induce acute effects in these organisms. However, Oehlmann *et al.* [233] suggest that effects at higher levels of biological organization may occur through subtle impacts on reproduction and development in vertebrates and invertebrates. They summarized the toxicological literature for types of responses in organisms exposed to BPA and concluded that invertebrates were generally more sensitive than vertebrates such as fish. In snails significant increases in reproductive effects and super-feminization have been observed, which is consistent with the proposed mode of action of BPA as an estrogen mimic. For example, in a 180-day study with *M. cornuarietis*, the EC10 for egg production (increase) was 13.9 ng/L, which is within the range of some environmental concentrations. Other invertebrates appear to be less sensitive. A NOEC of 1 mg/L was determined for reproduction in *D. magna*, and conflicting results have been reported in marine copepods, with some showing inhibition of larval development and others showing accelerated growth, including increased egg production at 20 µg/L. In *C. riparius*, emergence of second-generation individuals was delayed at concentrations as low as 78 ng/L. In fish, the majority of papers report feminization effects *in vivo* and expression of the vitellogenin protein, but at concentrations in the high µg/L and well above what is typically observed in aquatic environments. However, some studies have reported changes in circulating concentrations of some hormones at more environmentally relevant concentrations. Oehlmann *et al.* [233] cite one study on brown trout that reports impacts on sperm quality, a delay in ovulation, and inhibition of ovulation in the low µg/L BPA range, with the interpretation that this could lead to delayed breeding in less favorable periods, with potential impacts at the population-level for these fish. Based on available data, Oehlmann *et al.* [233] felt that BPA could be contributing to adverse reproductive outcomes in populations of exposed fish, but to date, no studies have shown this causally in the field. Staples *et al.* [257] summarized the chronic laboratory data (growth, reproduction and mortality) for this compound, developing species sensitivity distributions and estimating chronic predicted no effects concentrations (PNEC or the 5^{th} centile of the distributions), which are considered

protective of populations, communities and ecosystems. They determined PNEC values of 11 to 71 µg/L BPA and concluded, based on current environmental concentrations, that higher-level effects are not anticipated.

CONCLUSIONS

Global freshwater ecosystems have a long history of contamination from organic pollutants. While an enormous amount of research has been conducted to assess the impacts of organic contaminants on freshwater systems, much of this has been generated at lower levels of biological organization (organism and lower) and clear, cause-effect examples of contaminant-associated impacts at the population, community, and ecosystem level are generally rare (Table **1**).

Table 1: Relative state of current understanding, including cause-effect relationships, of the ecological impacts of the organic chemicals addressed in this review in relation to levels of biological organization. Relative rating based on evidence from laboratory, field, and cosm studies. XXX: clear evidence of impacts with some causal relationships established; XX: evidence of impacts but causal relationships not established or uncertain; X: possible evidence of impacts but causality not established; ___ no evidence of impacts.

Chemical	Sub-organism	Organism	Population	Community	Ecosystem
PCBs	XXX	XXX	XX	X	X
TCDDs/TCDEs	XXX	XXX	XX	X	X
PBDEs	XXX	XX	X	___	___
PAHs	XXX	XXX	XX	X	X
Pharmaceuticals	XX	XXX	X	___[1]	___
Bisphenol A	XXX	XXX	X	___	___
Nonylphenol	XXX	XXX	XX	X	X

[1] There is some evidence that community-level effects could occur in microbial communities

Our present understanding of how freshwater ecosystems respond to contaminants is largely based on work with persistent, bioaccumulative legacy chemicals (e.g. PCBs, dioxins, and PAHs) and the information derived from this work has proved essential in developing protective regulatory criteria and implementing ecosystem-based management strategies to mitigate effects. However, there is much research that remains to be done. For example, future research should focus on the potential impacts of more recent chemical classes (e.g. PBDEs, PFSs, and pharmaceuticals), whose physicochemical properties, environmental behavior, and potential impacts on aquatic ecosystems, has not been fully elucidated. In addition, since contaminants rarely occur individually in the environment, there is a need to better evaluate the impacts of chemical mixtures in aquatic systems and potential risks that result from exposure to them. Like individual chemicals, the historical focus for chemical mixtures assessment has been at lower levels of biological organization and there is little information about their effects at the population or community level. The potential effects of mixtures should be considered in the context of cumulative impacts, with emphasis on interactions between both chemical and non-chemical (e.g. nutrients, sedimentation, *etc.*) stressors. In terms of population and community-levels assessments, one of the ideal tools to undertake such studies is model aquatic ecosystems such as microcosms or mesocosms as these facilitate evaluation under "close-to-field" conditions, including direct and indirect effects, both of which may be critical aspects to quantify the fate and effects of organic chemicals in freshwater systems. Finally, greater resources are needed for chemical and biological monitoring of aquatic systems for the purpose of assessing trends in exposure to, and impacts from, chemicals to provide a stronger foundation on which to support regulatory and research initiatives.

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