Effects of Pollution on Marine Organisms

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ABSTRACT: This review covers selected 2014 articles on the biological effects of pollutants and human physical disturbances on marine and estuarine plants, animals, ecosystems and habitats. The review, based largely on journal articles, covers field and laboratory measurement activities (bioaccumulation of contaminants, field assessment surveys, toxicity testing and biomarkers) as well as pollution issues of current interest including endocrine disrupters, emerging contaminants, wastewater discharges, dredging and disposal, etc. Special emphasis is placed on effects of oil spills and marine debris due in part to the 2010 Deepwater Horizon oil blowout in the Gulf of Mexico and the 2011 Japanese tsunami. Several topical areas reviewed in the past (ballast water and ocean acidification) were dropped this year. The focus of this review is on effects, not pollutant fate and transport. There is considerable overlap across subject areas (e.g. some bioaccumulation papers may be cited in other topical categories). Please use keyword searching of the text to locate related but distributed papers. Use this review only as a guide and please consult the original papers before citing them.

KEYWORDS: Tissue residues, toxicity, bioaccumulation, biomagnification, biomarkers, sediment quality, ecological risk assessment, endocrine disrupters, nano particles, POPs, PCBs, PAHs, PBDEs, radionuclides, pharmaceuticals, personal care products, trace metals, pesticides, biomarkers, marine biocides, oil spills, dispersants, sewage, debris, dredging, eutrophication, human disturbance, Arctic, Antarctic.

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Syntheses and Overviews

Understanding the effects of chronic urban contamination on the longterm productivity of anadromous fish populations has been a challenge, requiring synthesis of habitat, contaminant and fish population patterns and trends. Meador (2014) published a synthesis of information on multidecadal-long smolt-to-adult return (SAR) rates of several dozen hatchery-released Chinook salmon (*Oncorhynchus tshawytscha*) and Coho salmon (*O. kisutch*) populations, comparing return rates with physical and chemical charachteristics of a wide range of urbanized to rural estuaries in southern, central and northern Puget Sound, Washington (USA). The study included existing and historical data on contaminant loadings of estuariene sediments, biota, and in salmon. Overall return rates of Chinook salmon, whose juveniles have long residence times in their estuaries, were significantly lower than for Chinook from less-contaminated rural estuaries whereas Coho salmon, whose juveniles run to sea rapidly showed no urban vs. less human-influenced differences in return rates. The author concluded that chemical contamination, marked by PCBs, has been a significant factor in the depressed Chinook return rates, additive to impacts from habitat disruption and other factors.

Michel and Rutherford (2014) conducted a detailed synthesis of historical information on the impacts and recovery rates of oiled marshes marshes. The review covered 32 cases of oil spills field experiments finding that in most cases recovery of marsh systems occurred within two growing seasons. Recovery was slowest (many growing seasons) in marshes subject to cold climates, heavy oiling and aggressive cleanup. Detailed response recommendations are provided.

Using worldwide tissue data collected as early as the mid-1970s, Law (2014) reviewed longterm trends of synthetic organic chemical contamination of marine mammals. Concentrations of organochlorine pesticides, PBDE's and butyltins have been slowly decreasing but also appear stalled and several are still at concentrations of considerable concern in killer whales and bottlenose dolphins. Data is insufficient to detwermine trends for perflorinated compounds, dioxans, furnans and related compounds.

Bioaccumulation and Biomagnification

Tabular Data. Table 1 lists examples of the concentration of contaminants in tissues of marine plants and animals in 2014. Data are presented alphabetically by element then by organic compound and arranged by geographical area, concentration and author. Following are reviews of selected additional papers that report on body burdens, bioaccumulation and biomagnification.

Metals. The interspecific variations in 15 trace element accumulation in 26 species of green, brown and red seaweeds were measured in collections in the Aegean Sea (Malea et al. 2014). Brown algae showed higher concentrations of arsenic and strontium than the other elements. Finely branched seaweeds had higher concentrations of cadmium, cobalt, copper, manganese and vanadium than species with broad thalli. There were species differences, for example, the brown algal species *Cystoseira* had the highest concentration of arsenic, the red species *Ceramium* with manganese, and the red *Polysphonia* and green *Ulva* had the highest amounts of cadmium and selenium. It was concluded by Malea et al. (2014) that the structure of the species of algae was the determining factor in the bioaccumulation of the element. The red seaweed *Gracilaria lemaneiformis* is an important plant used for food and for production of pharmaceuticals. Laboratory experiments were conducted to measure the bioaccumulations of cadmium, copper and lead (Wang, Wang and Ke, 2014). Concentrations of copper and cadmium in tissues were higher when the two elements were together compared to the elements alone. The results indicated that bioaccumulation does occur and that cultivation of the alga should be in clean sea water.

Kalantzi and associates (2014) measured the uptake of 28 elements in polychaetes, amphipods, decapods, molluscans and echinoderms at increasing distances from fish farms in Greece. Fish were also measured for bioaccumulation of the same elements. The invertebrates accumulated lower amounts of most elements than those measured in the sediments except arsenic, phosphorus, sodium, cadmium and zinc. Tolerant species, such as most of the species of polychaetes, accumulated higher concentrations of many of the 28 elements studied than the more sensitive species in different animals groups. Biomagnification of mercury was noted in fish feeding on benthic invertebrates in the vicinity of the fish cages.

A stable nitrogen isotope was used to document the biomagnification pathway of mercury and selenium occurs in fish (Jones et al. 2014). Results indicated that mercury but not selenium was taken up by fish feeding upon the animals from the local area. A risk assessment study was conducted in a marine bay in China to determine if butytin used in biofouling control would bioaccumulate in organisms used for food. It was found that few if any harmful compounds would occur in humans from consumptions of the cultured fish (Hu et al., 2014). Sublethal concentrations of cadmium, lead and a combination of the two elements were measured on mussel gill tissue (Poynton et al. 2014). Results of gene ontology identified several biological processes affected including nucleotide phosphate synthesis. These responses served as early indicators of stress which can lead to adverse physiological effects (Poynton et al., 2014).

Nanoparticles. The fate and effects of nanoparticles on a clam and polychaete were reviewed by Mouney et al. (2014). Data were analyzed to determine which route of exposure influenced bioaccumulation. Soluble nanoparticles attached to seawater molecules which were taken up in both species. The effect of cadmium sulfide nanoparticles on the biochemical and behavior responses to the polychaete *Hediste divericolor* was measured by Buffet et al. (2014a). Both solutions of cadmium and cadmium nanoparticles were accumulated by the polychaete. Behavior was measured by the number of body undulations. Body movements were impaired when exposed to cadmium nanoparticles compared to control. An experimental study with mussel, *Mytilus galloprovincialis,* was conducted by Balbi et al. (2014) to determine if manoparticles, as n-TiO^2, and cadmium together would induce measureable stress responses (biomarkers) without toxicity. The co-exposure of n-TiO^2

and cadmium resulted in effects on lysomal biomarkers and embryoni c development. However the toxicants did not result in an increase in adverse effects.

Tian et al. (2014) studied the impact of titanium dioxide nanoparticles in the presence and absence of phenanthrene under laboratory conditions. Bioaccumulations of $nTiO^2$ occurred in the presence of phenanthrene but not by itself. This difference is explained by the ingestion of the large molecule of $nTiO^2$ - phenanthrene complex by the clam. The toxicity and bioaccumulation of citrate and polyvinylpyrolides coated with silver nanoparticles were investigated using amphipods, mysids and polychaetes (Wang et al. 2014). These compounds were not toxic to the amphipods and mysids after a seven day exposure. However bioaccumulation occurred in the polychaete *Nereis virens*.

Synthetic Organic Chemicals. The uptake of PCBs and PBDEs in benthic fauna was measured near two marine sewage outfall, urban harbors, and a reference site (Burd et al. 2014). PBDEs were more readily taken up by biota and PCBs. However organic enrichment near waste discharges enhanced the uptake of PCBs (Burd et al. 2014). Accumulations of endrocrine disrupturing chemicals (EDCs) was measured in the polychaete *Paraprionospio* sp. from Osaka Bay, Japan (Nurulnadia et al., 2014). The polychaete accumulated varying concentrations of nonylphenol from 1.46 to 4.41 µg/g wet weight. Concentrations were greater in the polychaete than sediment indicating bioaccumulation in the worms (Nurulnadia et al., 2014).

An evaluation was made of the effect of diuron, a photosynthetic inhibitor, used in sugar fields near the Great Barrier Reef, Australia (Holms et al. 2014). In a 12 year review it showed that this pesticide affected coralline algae and photosynthetic cells in corals.

Handoh and Kawai (2014) divised a model to determine mass mortality of marine mammals. The mortality data on 33 species of marine mammals were compiled and analyzed with respect to the global concentration of PCBs in fish. Based on this model , hot spots were identified for marine mammal mortality as the Mediterranean Sea and north-western Europe. See Law (2014) for additional information on this subject.

Hydrocarbons. A laboratory experiment was employed to determine if three species of fish, one decapod crustacean and a mussel ingested oil droplets (Viaene et al., 2104). Selection of the test organism was based on feeding method: carnivores or filter feeders. Only the mussel took in the droplets indicating that filter feeding was the determining factor (Viaene et al., 2104). The benthic ecosystem in the Chukchi Sea was reviewed because of the potential for oil and gas exploration. The concentration of aliphatic hydrocarbons, polycylic hydrocarbons, and 17 metals were at background levels except where drilling was done earlier (Dunton et al. 2014). There was no evidence of bioaccumulation of these substances above background levels in biological benthic biomonitors such as crustaceans and echinoderms (Dunton et al., 2014).

Radionuclides. The uptake of polomium-210 was measured in muscle tissues and organs in predatory fish taken off Río de Janeiro by Mársico et al. (2014). There were significant differences in the ^{210}Po in the two species of fish. There were also differences of radioactivity in the different organs with the highest concentrations in the digestive system. A radionuclide transfer model employed the known data on uptake of inorganic in marine organisms for use in a planned disposal site in the Baltic Sea (Konovalenko et al., 2014). The risk assessment utilized data known for grazers, benthos, zooplankton and fish. It showed that the data on organisms is in good agreement, but many elements (i.e. 26) need to be used in addition to just cesium and strontium (Konovalenko et al., 2014).

The bioaccumulation of eight radioactive elements was conducted under laboratory conditions using the red alga *Polysiphonia fucoides (* Zalewska, 2014) After the initial exposure, additional uptake was linear regardless of the concentration of the isotope. Once the maximum concentration was reached,g a significant decline was noted in the isotopes of manganese, cobalt, cesium and americium (Zalewska, 2014).

Chemical Toxicity and Testing

Reviews below are organized by media or contaminant. Reviews of oil and dispersant toxicity follow oil spill sections at end of this paper.

Scientists continued to adapt existing methods and develop new methods to improve capabilities to quantify, analyze, and determine toxicity of various chemicals in the estuarine and marine organisms.

New Methods and Endpoints. Howe et al. (2014a) described an 8-d sublethal bioassay with sea anemone (*Aiptasia pulchella*) pedal lacerates. They exposed the pedal lacerates to Cd, Co, Cu, and Zn and observed effects on development to the juvenile stage (EC_{50}s were 55 µg/L, 262 µg L, 5 µg /L, and 269 µg /L for Cd, Co, Cu, and Zn, respectively). Howe et al. (2014b) used 28-d sublethal tests that observed asexual reproduction of symbiotic sea anemones (*Aiptasia pulchella*) to assess the toxicity of metals. They found 50% reductions in asexually reproduced juveniles (EC_{50}) at 14 µg/L for Cu, 63 µg/L for Zn, and 107 µg/L for Ni. Faimali et al. (2014) investigated the use of ephyra of jellyfish (*Aurelia aurita*) as a model organism. They examined the influence of different culture parameters (e.g., temperature, photoperiod) on behavioral endpoints and exposed the organisms to Cd nitrate and sodium dodecyl sulfate (SDS). They thought the ephyrae of jellyfish showed promise as experimental organisms. Renzi et al. (2014) exposed diatoms (*Phaeodactylum tricornutum*) to Zn, Cu, and dodecylbenzenesulfonic acid sodium salt in accordance with the AlgalToxkit protocol to quantitatively compare responses with growth rate inhibition tests morphological (biovolume) and physiological (chlorophyll-α, phaeophytin ratio) endpoints. The results showed that both biovolume and the photosynthetic complex are sensitive sub-lethal endpoints of exposure.

In an effort to improve the medaka (*Oryzias latipes*) embryo-larval sediment contact assay, Le Bihanic et al. (2014) developed a reference exposure protocol with artificial sediment that was specifically designed to limit natural sediment composition uncertainties and preparation variability. They tested and validated the artificial sediment using fluoranthene, benz[α]anthracene, and benzo[α]pyrene and observed various developmental end points.

Deep Sea Biota.Concerned with the advent of industrial activities in the deep sea, Mestre et al. (2014) described and emphasized the need for environmental risk assessments based on information generated from ecotoxicological trials that mimic, as close as possible, the deep sea environment, with emphasis to high hydrostatic pressure.

Metals. Oliveira et al. (2014) used solid phase sediment bioassays with copepods *Tisbe biminiensis* to assess the toxic effects of the sediments and trace metal analyses to measure trace metal concentrations (Cr, Zn, Mn, Fe, Cu, Pb, Co, and Ni) in the sediments from an urban mangrove patch in Brazil. They reported lethal and sublethal effect concentrations. Taylor and Maher (2014) used microcosms in a 28-d exposure of bivalves (*Tellina deltoidalis*) to selenium-spiked sediments (0, 5 µg/g, and 20 µg/g), and observed reduced antioxidant capacity, which corresponded with increased lipid peroxidation, lysosomal destabilization, and micronuclei frequency. Exposure-dose-response relationships were demonstrated for *T. deltoidalis* exposed to selenium spiked sediments. Dolores-Basallote et al. (2014) exposed amphipods (*Ampelisca abdita*) to Gulf of Cadiz sediments that had been subjected to several pH treatments to study the effects of CO_2-induced acidification on sediment toxicity. The results showed that CO_2-related acidification lead to lethal effects in amphipods and mobility of metals. Canário et al. (2014) established baseline sediment toxicity levels by testing and analyzing Canadian Arctic and sub-Arctic sediments collected in 2005 from eight locations. Sediment elutriates were tested with ARTOXKIT M, Microtox liquid phase, and ROTOXKIT M toxicity assays, and whole sediment was tested with the Microtox solid phase assay procedure. The sediments were considered nontoxic, although some measures of toxicity were measurable. Hoang and Rand (2014) conducted short term and longer term whole sediment acute toxicity studies with amphipods (*Ampelisca abdita*) and clams (*Mercenaria mercenaria*) to examine seven field collected sediment samples from the St. Lucie River (Florida) and found no significant effect on survival and an inverse relationship between dry weight of the tested organisms and Cu and Zn concentrations in sediments and organisms.

Hariharan et al. (2014) conducted acute and chronic toxicity tests to examine the adverse effects of Pb on mussels (*Perna viridis*). In the chronic exposure, they correlated Pb concentrations with adverse effects as shown in changes in biochemical enzymes, catalase, glutathione, glutathione *S*-transferase, and lipid peroxides.

Anti Fouling Biocides. Several scientists investigated anti-fouling compounds, aquaculture pesticides, and other pesticides. Fox et al. (2014) exposed 7-d old amphipod (*Corophium volutator*) neonates to medetomidine, an anti-fouling compound, in 28-d and 76-d sediment studies. They observed mortality at 32 µg/kg. Arrhenius et al. (2014) described a bioassay that examined the effects of biocides on the initial settling and establishment of photoautotrophic biofilms, including indigenous fouling organisms. Using this bioassay, Arrhenius et al. (2014) ranked the efficacy (based on EC_{98} values) of select antifoulants from most to least efficacious: Cu pyrithione> TPBP (triphenylborane)> DCOIT (4,5-Dichloro-2-octyl-1,2-thiazol-3-one)> tolyfluanid> Zn pyrithione> medetomidine> $Cu^{2+>}$. Bao et al. (2014) examined the combined toxicity of zinc pyrithione and Cu on copepods (*Tigriopus japonicus*) using 96-h acute toxicity tests with adult copepods and 21-d chronic full life-cycle tests with nauplii. Zinc pyrithione has been reported to be easily trans-chelated to Cu pyrithione in the presence of Cu. Their results showed that Zn pyrithione and Cu exhibited strong synergistic toxic effects in both acute and chronic tests.

Li et al. (2014) examined the toxicity-temperature relationship using 96-h LC_{50} tests with fish (*Oryzias melastigma*) and copepods (*Tigriopus japonicus*) and the 24-h LC_{50} test with rotifers (*Brachionus koreanus*). The toxicants were Cu and three biocides [dichlorophenyltrichloroethane (DDT), triphenyltin chloride (TPTCl), and copper pyrithione (CuPT)]. The toxicity for fish peaked at 20°C for Cu and TPTCl and at 25°C for DDT and CuPT. The LC_{50} values for copepods and rotifers showed an inverse relationship with temperature across all the toxicants used. Yi et al. (2014) examined the effects of triphenyltin chloride on molecular, individual, and population responses of copepods (*Tigriopis japonicus*). In a full life-cycle exposure, 1.0 µg/L elicited a delay in development and a reduction in population growth. At 0.1 µg/L, the sex ratio changed to a male-biased population. They also observed the inhibition of transcriptional expression of glutathione *S*-transferase genes and inhibition of retinoid X receptor mRNA expression.

Wang, Shi, Yang, Han ans Zhou (2014) used fish (*Brachydanio rerio*) and green algae (*Selenastrum capricornuttum*) in acute and growth inhibition tests, respectively, to assess capsaicin (an active substance for ship anti-fouling systems). They calculated the Predicted No Effect Concentration (PNEC) to be 4.9 x 10-4 mg/L, which was one to two orders of magnitude above the average PEC for OEECD-EU commercial harbors and marinas, respectively.

Wang, Zhao, Yang, Han, Long and Zhou (2014) used available data to develop risk assessments for active anti-fouling substances used in China.

Pesticides. The toxicity of four anti-lice pesticides, used in salmon aquaculture (deltamethrin, cypermethrin, hydrogen peroxide, and azamethiphos), to copepods was examined in 1-h exposures with lethality and feeding endpoints determined 5-h post-exposure using staining techniques. EC_{50}s ranged from 30- to 117-fold, 13- to 51-fold, and 120- to 460-fold dilutions of the respective aquaculture treatments for deltamethrin, cypermethrin, and hydrogen peroxide, respectively. No effects on copepods were observed at 5-times the aquaculture treatment concentrations for azamethiphos (Van Geest et al., 2014a). Van Geest et al. (2014b) used the amphipod *Echinogammarus finmarchicus* to test the toxicity of the pyrethroid-based anti-sea lice pesticides, deltamethrin and cypermethrin in aqueous and sediment exposures. In 1- and 24-h water-only exposures, mortality and immobility were observed at 6.7-70 ng/L and 20-220 ng/L for deltamethrin and cypermethrin, respectively, suggesting that amphipods could be exposed to concentrations of pyrethroids sufficient to cause adverse effects in effluent plumes from salmon aquaculture where treatment for ectoparasitic sea lice is occurring. Tucca et al. (2014) examined the effects of three antiparasitic pesticides (cypermethrin, deltamethrin, and emamectin benzoate) on amphipods (*Monocorophium insidiosum*) in 10-d whole sediment bioassays. They observed effects on glutathione *S*-transferase and thiobarbituric acid reactive substances. *M insidiosum* was sensitive to antiparasitic pesticides concentrations at µg/kg in sediments. Van Geest et al. (2014c) examined the effects of the anti-sea lice pesticide deltamethrin on *Nereis virens* (polychaete) exposed via water and sediments. Sublethal effects related to burrowing behavior and worm condition were observed at 11 µg/g. DeLorenzo et al. (2014) exposed 7-d old mysids (*Americamysis bahia*) and larval and adult shrimp (*Palaemonetes pugio*) to pyrethroid insecticides (lambda-cyhalothrin, permethrin, cypermethrin, deltamethrin, and phenothrin) in 96-h LC_{50} bioassays. Acute toxicity occurred at low nanogram per liter concentrations for some of the pyrethroids tested.

Herbicides. Yusof et al. (2014) exposed fertilized fish eggs (*Oryzias javnicus*) to high concentrations (100 to 500 ppm) of a glyphosphate-based herbicide (Roundup®). Survival and hatching percentages decreased as glyphosphate concentrations increased. Teratogenic effects were observed. Hwang et al. (2014) examined the toxicity of triclosan to gamete viability, fertilization, and embryogenesis in sea urchins (*Strongylocentrotus nudus*). The LOEC for embryogenesis was 0.33 µM.

Personal Care Products. There is continued concern about the effects of personal care and health products on marine organisms. Several investigators examined the effects of antibiotics, antidepressants, and beta blockers. Seoane et al. (2014) investigated the toxic effects of three antibiotics (chloramphenicol, florphenicol, and oxytetracycline) on algae (*Tetraselmis suecica*) and found that photosynthetic parameters, such as chlorophyll a cellular content and auto fluorescence, were altered after 24 h of antibiotic exposure. Gomiero and Viarengo (2014) investigated the effects of Cu and oxytetracycline, separately and in combination, at different temperatures on the survival, replication, endocytosis rate, and lysosomal membrane stability in the ciliated protozoa *Euplotes crassus*. The two toxicants elicited very different and complex trends under the various thermal conditions. Johansson et al. (2014) examined the toxicity of two antibiotics, ciprofloxacin and sulfamethoxazole, to the bacterial and algal species on natural biofilms from the coast of Sweden. They used a 4-d semi-static system to expose the biofilms, and found that sulfamethoxazole exposure leads to a general decrease in carbon source utilization and ciprofloxacin exposure leads to a re-arrangement of the carbon-utilization pattern in the bacteria species

Minguez et al. (2014) assessed the ecotoxicity of three antidepressants (fluoxetine, sertraline, and clomipramine) using a battery of tests/species [algal growth inhibition (*Skeletonema marinoi* and *Pseudokirchneriella subcapitata*), crustacean immobilization (*Artemia salina* and *Daphnia magna*), development and adult survival (Hydra attenuate), oyster embryogenesis and metamorphosis (*Crassostrea gigas*), and *in vitro* assays on abalone hemocytes (*Haliotis tuberculata*). EC_{50} values ranged from 43 to 15,600 µg/L for fluoxetine, 67 to 4,400 µg/L for sertraline, and 4.70 to 100,000 µg/L for clomipramine. The algae and embryo-larval stages of the oyster were the most sensitive of the taxa tested. Maszkowska et al. (2014) conducted ecotoxicological evaluations on three beta-blockers (propanolol, metaprolol, and nadolol) using bacteria (*Vibrio fischeri* and *Arthrobacter globiformis*), algae (*Scenedesmus vacuolotus*), and duckweed (*Lemna minor*). They found that propanolol and metaprolol were harmful to algae.

Paredes et al. (2014) examined the toxic effects of four chemical UV filters (4-Methylbenzylidene-camphor, Benzophenone-3, Benzophenone-4, and 2-Ethylhexyl-4-methoxycinnamate) on algae (*Isochrysis galbana*), mussels (*Mytilus galloprovincialis*), sea urchins (*Paracentrotus lividus*) and mysids (*Siriella armata*). They found 2-Ethylhexyl-4-methoxycinnamate and 4-Methylbenzylidene-camphor to be the more toxic of the chemicals tested, with the algae being the most sensitive of the tested species

Multiple Products. Fabbri et al. (2014) demonstrated that acute 48-h mussel (*Mytilus galloprovincialis*) toxicity tests can be used to evaluate the effects of different model compounds representative of endocrine disrupting chemicals (nonylphenol, bisphenol A), brominated compounds (tetrabromobisphenol A), perfluorinated compounds (perfluorooctanoid acid and perfluorosulphonate), and pharmaceuticals (ibuprofen, diclofenac, and benzafibrate) in a wide concentration range (0.01 – 1000 µg/L). Rhee, Jeog, Kim and Lee (2014) reported inhibition of growth of *Barachionus koreanus* to a suite of pharmaceuticals.

Household Cleaning Products. The 96H LC50s of eight household cleaning products, ranging from baby shampoo to dishwashing detergents, to mysid shrimp (*Americamysis bahia*) and silversides fish (*Menidia beryllina*) was reported by Word et al. (2015). For the fish, 96H LC50s ranged from 5.4 to 592 mg/L and for the mysid shrimp from 12.4 to 413 mg/L. Liu and Zhang (2014) experimentally compared 25 halophenolic and haloaliphatic disinfection byproducts using algae (*Tetraselmis marina*) as the test organisms. Halophenolic disinfection byproducts were generally more toxic than the commonly known haloacetic acids.

Nano Materials. Choi et al. (2014) used algae (*Pseudokirchneriella subcapitata*), daphnia (*Daphnia magna*), and bacteria (*Vibrio fischeri*) to examine the toxicity of water-solubilized aminoclay nanoparticles. The EC_{50} for algal growth was 1.29 mg/L and 0.26 mg/L for cell yield, while the NOEC for *V. fischeri* was 25,000 mg/L and the LOEC (48-h) for *D. magna* was 100 mg/L. Castro-Bugallo et al. (2014) examined the effects of two metal oxide nanoparticles [zinc oxide (ZnO) and yttrium oxide (Y_2O_3)] on three microalgae (*Phaeodactylum tricornutum, Alexandrium minutum*, and *Tetraselmis suecica*). They tested the effects on growth, carbon content, carbon-to-nitrogen ratio, chlorophyll fluorescence, and reactive oxygen species production. Pronounced differences were observed between species to the test materials highlighting the importance of analyzing diverse groups of microalgae and various physiological effects. Park et al. (2014) assessed the effects of salinity (5‰ - 35‰) on Cu, Zn, and CuO nanoparticles and ZnO nanoparticles on copepods (*Tigriopus japonicus*) in 96-h exposures. They observed no differences in acute toxicity at different salinities. Wang et al. (2014) investigated the toxicity, bioaccumulation, and biotransformation of citrate and polyvinylpyrrolidone coated Ag nanoparticles in *Nereis virens* (polychaete), *Ampelisca abdita* (amphipod), and *Americamysis bahia* (mysid) via marine sediment exposure. The results showed that the silver nanoparticle surface capping agents influenced Ag uptake, biotransformation, and/or excretion, and demonstrated the accumulation and speciation of Ag nanoparticles in *Nereis virens*. Minetto et al. (2014) discussed the complexity of experimental exposure scenarios for titanium oxide nanomaterials in the marine environment: providing statistical information about different matrices, organisms, and nanomaterials, and comparing ecotoxicity effects of titanium oxide nanomaterials.

Other Commercial Compounds. Ventura et al. (2014) provide ecotoxicological data for ten cholinium-based salts and ionic liquids (chollinium bicarbonate, cholinium bitartrate, cholinium chloride, cholinium acetate, cholinium diihydrogenophosphate, cholinium dihydrogenocitrate, cholinium salicylate, benzyldimetthyl(2-hydroxyethyl)ammonium chloride, cholinium propanoate, and cholinium butanoate) to gain insight on the toxicity mechanism of these compounds. They found that not all the cholinium tested is harmless to bacteria (*Vibrio fischeri*), and that the cholinium exhibits a diffferent machanism of toxicity compared to the imidazolium ionic liquids previously described.

Sediment Quality Guidelines. Kwok et al. (2014) summarized the key findings of an expert group convened to discuss the important scientific and regulatory challenges with developing sediment quality guidelines. They also identified areas of scientific research that are needed to improve sediment quality assessment. Diepens et al. (2014) produced a review paper that describes sediment toxicity tests for microorganisms, macrophytes, benthic invertebrates, and benthic communities. They presented recommendations and identified knowledge gaps and priorities for further research. Field and Norton (2014) describe approaches to improve the performance of empirical models developed from a large nationwide data set to predict sediment toxicity from chemistry from regional applications. Their study suggests that calibrating nationwide models to a regional data set may be a more efficient and effective approach for improving model performance than developing region-specific models.

PAHs. Mu et al. (2014) exposed early life stages of fish (*Oryzias melastigma*) to phenanthrene and retene (7-isopropyl-1-methylphenanthrene) and found retene to be more toxic. Both phenanthrene and retene caused developmental malformation of embryos with phenanthrene affecting the peripheral vascular system and retene affecting cardiac tissues. Wang, Wang, Mu, Wang, Yao and Lin (2014) used existing toxicity data for Chinese marine species to derive predicted no-effect concentrations for three polycyclic hydrocarbons (phenanthrene, pyrene, and benzo(α)pyrene). Pastore et al. (2014) examined the kind of nuclear damage found in fish (*Sparus aurata*) hepatocytes continuously exposed to sublethal doses (10 μg/mL – 1pg/mL) of benzo[α]pyrene (B[α]P) for 24 and 72 h. They observed apoptosis induction and nuclear abnormalities by immunofluorescence analysis and found no evidence of a threshold dose below which B[α}P was found not to be genotoxic in sea bream cultured hepatocytes.

Biomarkers

Biomarker Reviews. Several biomarker reviews were published in 2014. Regoli and Giulani (2014) wrote a very useful review of oxidative stress biomarkers in marine organisms. They covered what each biomarker does, how it is regulated, and how it tends to respond to given chemical pollutants. They also discussed the reasons that researchers often find mRNA levels do not correspond to catalytic activities of antioxidant enzymes, including differences between transcriptional and post-translational regulation. Bolognesi and Cirillo (2014) reviewed the use of the comet assay and micronucleus test to measure genotoxicity in *Mytilus* spp. They found that both methods were useful, and they demonstrated differing sensitivity to contaminant classes, possibly because the comet assay provided detection of recent exposure, whereas the micronucleus test could reveal accumulated damage during the lifespan of the organism. It was important to take into account the effects of other factors besides chemical pollution, and they recommended using large sample sizes, caged mussels, standardized deployment and sampling protocols to minimize variability in response. A review of environmental stress proteomics studies by Tomanek (2014) found that comparing responses of closely related species to population responses provided information on the cellular processes that set the environmental tolerance limits of a given species. Rodrigues and Pardal (2014) reviewed studies with the green crab *Carcinus maenas* and developed a list of the most suitable biomarkers to detect response to classes of contaminants, for example the use of the ethoxyresorufin-O-deethylase (EROD) assay with hepatopancreas tissue for polycyclic aromatic hydrocarbon (PAH) and polychlorinated biphenyl (PCB) exposure. They found that confounding factors for data interpretation included gender, size, morphotype, nutritional status, historical contaminant exposure, temperature and salinity. Lehtonen et al. (2014) presented recommended biological effects indicators for the Baltic Sea based on several region-wide studies. These included lysosomal membrane stability (LMS) in fish, bivalves or amphipods to measure general stress caused by a range of contaminants; induction of micronuclei in fish, bivalves or amphipods for effects of genotoxic contaminants; embryo aberrations in eelpout or amphipods for reproductive success impairments; fish disease index based on externally visible fish disease, macroscopic liver neoplasms and liver histopathology for general health status; imposex in gastropods for tributyltin effects; and PAH metabolites in fish for PAH exposure. Other methods were placed on the candidate list, including intersex or vitellogenin (VTG) in male fish to measure endocrine disruption, acetylcholinesterase (AChE) activity for neurotoxicity, and EROD for biotransformation.

Field Studies: Molluscs. Tsangaris et al. (2014) placed caged mussels (*Mytilus galloprovincialis*) for one month periods both at and near a disposal site for dredged river sediment in Greece. They found that sediment contaminant loads and Microtox® toxicity were highest at a location within the dumping zone and persisted even after dumping was terminated. However, PAH and aliphatic hydrocarbon levels in mussel tissues were elevated at all sites, even one outside of the immediate dumping zone, and the elevated contaminant load was accompanied by significant decreases in condition index (CI), scope for growth (SFG), glutathione-*S*-transferase (GST) activity, and catalase (CAT) activity but had no effect on lipid peroxidation (LP), metallothionein (MT) levels or AChE activity in digestive glands. Contaminant load and biomarker effects decreased after termination of dumping, indicating the water column effects, though more widespread during dumping, were more transient than possible benthic effects at the dump site. Mussels (*M. galloprovincialis*) transplanted for 4 weeks in two harbors on the Atlantic coast of Spain had induced gill GST activity that correlated with the sediment chemical pollution index (CPI), mussel bioaccumulation index (MBI), benzo(a)pyrene (BaP), and Cu in mussel tissue; induced gill glutathione peroxidase (GPx) activity that correlated with MBI and BaP; and inhibited gill AChE activity that was negatively correlated with sediment CPI and mussel Cu concentration (Vidal-Liñán et al. 2014). Bellas et al. (2014) combined effects on gill GST, GPx and AChE activities and SFG metrics into an Integrated Biomarker Response (IBR) to examine possible effects of pollutant bioaccumulation in wild *M. galloprovincialis* collected from 40 locations on the N-NW coast of Spain. The highest IBR values were correlated with Σ dichlorodiphenyl trichloroethane (DDT) and Σ hexachlorocyclohexane concentrations in mussel tissue. Lacroix et al. (2014) tested a series of potential reference genes to determine the best to use in a study looking at effects on mRNA biomarkers in caged (1 month) and wild mussels (*Mytilus* spp.) in the Bay of Brest, France. They found it was best to use an index of three of the most stable genes as a reference, which included *28s rRNA*, elongation factor (*ef*) *2* and *ef1α* in gills and α-tubuline (*αtub*), ribosomal protein L7 and actin (*act*) in digestive glands. The best biomarkers for differentiating between caged, wild, and reference mussels were cytochrome P450-3-like-2 (*cyp32*), *π-gst* and Cu/Zn-superoxide dismutase (*CuZn-sod*) in gills. Brenner et al. (2014) found that mussels (*M. edulis*) grown from larvae hanging in the water column offshore matured earlier in the year, had little to no parasites, but accumulated more metabolic concentrations of lipofuscin and neutral lipids compared to intertidal mussels. They also developed a lighter shell, which significantly skewed the CI compared to intertidal mussels, leading the authors to conclude that intertidal and offshore mussels should be separated into subgroups when conducting large scale environmental programs. The gastropod *Morula granulata* was collected from 9 intertidal sites off the coast of Goa, India and analyzed for DNA integrity using the time-dependent partial alkaline unwinding assay, and DNA damage using the comet assay (Sarkar et al. 2014). Both methods showed similar trends, with increased effects correlating with PAH concentration in sediments, and interestingly also nitrate, salinity and phosphate in the collection site water. Seabra Pereira et al. (2014) performed an extensive study with both mussels (*Perna perna*) caged for 3 months and oysters (*Crassostrea rhizophorae*) caged for 28 days in an industrialized estuary on the São Paulo Coast, Brazil. Mussels caged near two sewage outfalls had increased gill EROD, GST, GPx and dibenzylfluorescein dealkylase (DBF) activity, along with increased DNA strand breaks and gill histopathology and decreased LMS, which correlated with degraded macrobenthic community structure in the area. Similarly oysters caged at industrial areas had higher gill EROD, GST, DBF activity, DNA damage, LP and lower LMS which also correlated with degraded macrobenthic community structure. There was also an interesting association of decreased gill LMS with decreased normal development of the larvae from spawned caged mussels.

Field Studies: Crustaceans. Vasanthi et al. (2014) examined heavy metal bioaccumulation and histopathology in mud crabs (*Scylla serrata*) from brackish Pulicat Lake in India. Concentrations of Cu, Pb, Zn, Cd, Mn and Fe were all significantly elevated above those in reference crabs, and concentrations were higher in gills and hepatopancreas over muscle tissue. All three tissue types demonstrated histopathology, including lamellae abnormalities in gills, necrotic tubules and thickened basal lamina in the hepatopancreas, and atrophy and connective tissue damage in muscles. Jebali et al. (2014) placed caged Mediterranean crabs (*Carcinus maenas*) for up to 60 days in Téboulba Harbor, Tunisia, where PAH and metal concentrations exceeded sediment quality guidelines. Proteomic analysis of hepatopancreas tissue indicated time-dependent effects on proteins including chitinase, carboxypeptidase B, cathepsin L, cryptocyanin, heat shock protein 70 (HSP70), hemocyanin and monocarboxylate transporter. Ramos et al. (2014) collected gooseneck barnacles (*Pollicipes pollicipes*) once a season from three locations in north Portugal. Within season GST activity was for the most part elevated at Lavadores (municipal contamination) and Matosinhos (oil refinery contamination) over the reference in both the cirrus and peduncle. Cirrus LP was decreased at the two contaminated sites during all seasons, but peduncle AChE activity decreased at Lavadores only in winter and spring, and at Matosinhos in spring and summer. There were large increases in cirrus LP and GST activity and to a lesser extent peduncle GST activity in spring and summer across all sites, indicating a possible confounding factor in toxicity data interpretation.

Field Studies: Fish. Pre-spawning female grass goby (*Zosterisessor ophiocephalus*) were collected from five locations in Bizerte lagoon, Tunisia (Barhoumi et al. 2014). Significant positive correlations were found between liver EROD, LP, GST and CAT activities, and muscle tissue ΣPAHs, ΣPCBs and organochlorine pesticides. Muscle AChE activity was inversely correlated with muscle hexachlorobenzene and ΣDDTs, and no correlation was found between metals in muscle and any biomarker.

Dos Santos et al. (2014) examined correlations between liver butyltin concentrations and bile PAHs and liver biomarkers in marine catfish (*Cathorops spixii*) collected from Paranaguá Bay, Brazil. Lower EROD and GST activity, lower DNA damage and higher LP and histopathology were found in fish from the location where tributyltin was highest, versus the sight where bile PAHs and dibutyltins were highest. Lyons et al. (2014) examined effects of location, liver concentrations of coplanar PCBs, sex and age on CYP1A content and EROD activity in California round stingrays (*Urobathis halleri*) off southern California, USA. Males from the mainland had significantly greater EROD activity and CYP1A content than those from the relatively cleaner Santa Catalina Island, but EROD and CYP1A values for females were similar between the two locations, even though the PCB concentrations were much higher in the mainland females. Adult males had lower EROD activity than juveniles, females had a negative correlation between disk width and EROD activity, and both CYP1A and EROD were lower in females than males of all age classes. Podolska et al. (2014) performed an interesting study in the southern Baltic Sea looking at AChE activity in both cod (*Gadus morhua*) and the parasitic acanthocephalans (*Echinorhynchus gadi*) in their gastrointestinal tract. Female cod had lower AChE activity then males, and there was an inverse relationship between the cod and the *E. gadi* AChE activity. There were no site differences in the cod AChE activity, but there were in the *E. gadi* activity, indicating that the parasite may be more sensitive to environmental changes than the host.

Field Studies: Sea Turtles and Mammals. Guerranti et al. (2014) measured levels of porphyrins, intermediate metabolites in heme biosynthesis, in stranded loggerhead turtles (*Caretta caretta*) in Italy and tried to correlate them with bioaccumulation of several organic compounds. They found a positive correlation between liver perfluorooctane sulfonate (PFOS) concentrations and uroporphyrins. Lehnert et al. (2014) found that hematology markers in rescued abandoned grey seal pups (*Halichoerus grypus*) from the North and Baltic Seas did not vary between admission and release from a rehabilitation facility, but transcription of *hsp70*, aryl hydrocarbon receptor, aryl hydrocarbon receptor nuclear translocator, and to a lesser extent peroxisome proliferator-activated receptor alpha and cytokine interleukin-2 (*IL-2*) in the blood were all higher at admission than release, and lower in pups than adults. Polizzi et al. (2014) sampled Franciscana dolphins (*Pontoporia blainvillei*) incidentally caught by fisherman from an environmentally impacted estuary and a nearby open ocean area off Argentina, and found no correlation between renal or hepatic Cu and Zn and MT concentrations. Seven stranded sperm whales (*Physeter microcephalus*) in southern Italy were sampled by Marsili et al. (2014). Western blot revealed similar CYP1A1 and CYP2B levels in the skin, but CYP1A1 was higher than CYP2B in the liver. Cell cultures were developed from fibroblasts of the whales, and CYP1A1 was induced in the cell cultures by exposure to a mixture of Arochlor 1260, pp'DDT and pp'DDE, and both CYP1A1 and CYP2 were induced by a mixture of BaP and beta-naphthoflavone.

Laboratory Studies: Metals. Green mussels (*Perna viridis*) exposed to Pb for 30 days accumulated the metal several fold above control levels at all concentrations tested (0.008 to 0.109 mg/L) (Hariharan et al. 2014). Several biomarkers were measured in gill and whole body, but only CAT activity (decrease) and LP (increase) in gill and histology of the gill and adductor muscle showed effects at Pb concentrations below the survival LOEC of 0.026 mg/L. Lead caused microtubule cytoskeleton depolymerization and cell death in the seagrass *Cymodocea nodosa* at tissue concentrations of at least 18.33 and 20.24 µg/g dry weight, respectively (Malea et al. 2014). Toxicity appeared to correlate better with rate of Pb uptake rather than to total tissue Pb concentration. Won et al. (2014) identified genes for Cu/Zn-SOD1, Cu/Zn-SOD2, and Mn-SOD in the polychaete *Perinereis nuntia*, then exposed the polychaete to 50 µg/L As, Ni and Pb either singly or in combination for 48 hours. Lead alone and in combination with As enhanced expression of *Cu/Zn-sod1*. The mixture of As and Pb also enhanced *Mn-sod*, more so than Pb alone, As caused a gradual decrease of expression, and the combination of all three metals depressed expression. *Cu/Zn-sod2* was not as sensitive as the other two forms of SOD.

Manila clams (*Ruditapes philippinarum*) exposed to 4 µg/L Cd for 35 days then depurated for 35 days had depressed O:N ratios (a measure of energy source) from day 7 of exposure to 15 days after removal of Cd, and depressed SFG from day 21 of exposure to 7 days after removal (Zhao et al. 2014). Clams exposed to 40 µg/L Cd had depressed O:N ratio and SFG throughout the exposure and depuration, and SFG was negative for days 21 to 35 of exposure, indicating that the clams were unable to obtain enough energy to fuel their metabolic requirements. Chalkiadaki et al. (2014) exposed the bivalves *Mytilus galloprovincialis*, *Callista chione* and *Venus verrucosa* to 0.5 mg/L Cd for 20 days followed by 10 to 30 days depuration. Cadmium accumulation and MT levels were positively correlated in gill, mantle and body for *M. galloprovincialis*, in mantle and body for *C. chione* and gills for *V. verrucosa*. Cadmium accumulation was positively correlated with LP in *M. galloprovincialis* gills and mantle, and *C. chione* gills, mantle and body, but there was no correlation in *V. verrucosa* tissues. Cadmium accumulation was negatively correlated with AChE in *M. galloprovincialis* gills, mantle and body and *C. chione* mantle and body, but again there was no correlation in *V. verrucosa* tissues. Neither Cd tissue levels nor biomarkers returned to pre-exposure levels after depuration in any species.

Kim et al. (2014) exposed the copepod *Tigriopus japonicus* to up to 100 µg/L of Cu, Zn, Ag, As and Cd for 96 hours to determine effects on reactive oxygen species (as measured by production of dichlorofluorescein from 2',7'-dichlorodihydrofluorescein diacetate) and antioxidant enzyme activities. Cadmium was the least toxic increasing only GST, GPx and SOD activities, and Cu was the most toxic increasing ROS, GSH, GST, glutathione reductase (GR), GPx and SOD, and the toxicity of the other three metals fell in order of Zn>Ag>As. Transcription patterns of 9 HSP genes were also examined, Cu affected the most genes, and *hsp20* and especially *hsp70* were found to be the most sensitive genes to all five metals. *Heat shock protein 20* expression was also found to be sensitive to Cu as well as Ni in the diatom *Ditylum brightwelii*, but expression was not sensitive to bisphenol A, Arochlor 1016 or endosulfan (Lee, Guo and Ki 2014). Liu, Xiang amd Shao (2014) characterized a gene for globular C1q-domain-containing proteins (C1qDC), which are involved in immune response, in *Mytilus coruscus*. They found the gene was temporarily up-regulated upon exposure to pathogenic bacteria, perhaps indicating an acute-phase role in immune response. Exposure for 30 days to 20 µg/L Cu caused up-regulation by day 5, but the levels dropped to background by day 20, and 0.2 mg/L Cd only significantly increased expression on day 20 of the exposure. Morris et al. (2014) characterized the gene for Vacuolar-ATPase subunit A (fVHA-A) and a truncated, potentially non-functional modified RNA transcript (tVHA-A) in the alga *Fucus vesiculosus*, and also developed antisera for the proteins. Algae collected from sites with more metal contamination had higher fVHA-A protein content and *fVHA-A* expression than sites with lower contamination, and also showed the presence of tVHA-A protein and *tVHA-A* expression, where other sites had none. The concentration of the fVHA-A protein increased in algae exposed *in vitro* to 30 µg/L Cu over 11 days. In 300 µg/L Cu fVHA-A further increased, and tVHA-A began to appear after day 6, and gene expression patterns were similar. This may be the first example of an RNA editing event induced by environmental stress.

Laboratory Studies: Nanoparticles. Hu et al. (2014) exposed *Mytilus edulis in vivo* and their hemolymph *in vitro* to 400 to 1000 ppb CuO nanoparticles (NP) for 1 hour. In the *in vivo* experiment, the gill tissue contained two proteins that were targets of thiol oxidation (act and triosephosphate isomerase) and three that were targets for carbonylation (α-tub, tropomyosin and Cu/Zn-SOD), indicative of oxidative damage, and the mantle, connective tissue and digestive tubules had increased deposition of pigmented brown cells at all concentrations. All concentrations also increased LMS *in vitro*. The ragworm *Hediste diversicolor* and the clam *Scrobicularia plana* were exposed in a mesocosm for 21 days to 10 µg/L of either soluble Ag ($AgNO_3$) or Ag NP (Buffet et al. 2014b). Both forms induced oxidative stress, detoxification, apoptosis, genotoxicity and immunomodulation in both species, but GST, SOD, LP and burrowing rate in worms and capsase 3-like protein and burrowing rate in clams were more sensitive to soluble Ag. Nanoparticle Ag caused higher lysozyme activity in worms, and higher laccase-type PO protein levels and DNA damage in clams. Balbi et al. (2014) exposed *M. galloprovincialis* for 96 hours to 100 µg/L of Cd^{2+} and TiO_2 NP both alone and in combination to examine interactive effects. Nanoparticle TiO_2 increased NO production and serum lysozyme activity in hemocytes, but not in combination with Cd. Cadmium increased expression of *mt10* and *mt20* in hemocytes, but not in combination with TiO_2 NP. In the digestive gland additive effects were observed on LMS, and the mixture increased lysozyme and decreased toll-like receptor protein expression when neither toxicant alone had an effect. A similar study was performed by Canesi et al. (2014) examining the interaction of TiO_2 NP and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) in *M.*

galloprovincialis both *in vitro* and *in vivo*. There was an antagonistic effect between the two toxicants on phagocytic activity of hemocytes and efflux activities mediated by ABC transporters in gills both *in vitro* and *in vivo*, and synergistic effects on digestive gland lysosome/cytoplasm volume ratio and estrogen receptor transcription *in vivo*. Also, interestingly, TiO_2 NP also caused significantly higher TCDD to be accumulated in whole soft tissue than when the mussels were exposed to TCDD alone.

Laboratory Studies: Pharmaceuticals and Personal Care Products. Munari et al. (2014) exposed the clam *Venerupis philippinarum* to fluoxetine (a selective serotonin reuptake inhibitor (SSRI) and the active ingredient in the antidepressant Prozac®) for 7 days to examine effects on immune parameters and gill AChE activity. The lower concentrations tested (1 to 5 µg/L) decreased LMS and AChE activity, whereas the higher concentrations (25 to 625 µg/L) increased hemocyte proliferation, and at 25 µg/L only, total hemocyte count. The effects of 0.001 to 1 µg/L fluoxetine and another SSRI antidepressant, sertraline, on amphipod (*Echinogammarus marinus*) light-response behavior and transcription of light-response related genes were examined by Bossus et al. (2014). The amphipods were significantly more active in both light and dark phases, and recovery time from light stimulation was altered for up to 1 day of exposure to each SSRI, but by day 8 no significant differences were observed. On day 8 transcription of rhodopsin, arrestin and neurocan core protein in the head of the amphipod were all downregulated in the lower fluoxetine concentrations, but there were no significant effects in the sertraline exposure. Ribalta and Solé (2014) exposed liver microsomes of six fish species to up to 100 µM of diclofenac (analgesic), fluoxetine, gemfibrozil (lipid regulator), galaxolide (fragrance), nonylphenol (detergent degradation product) or triclosan (antibacterial agent) to examine effects on cytochrome P450 activity. All compounds depressed EROD activity in most fish species. 7-benzyloxy-4-trifluoromethylcoumarin-*O*-debenzyloxylase activity was depressed in the most species by diclofenac, and was significantly depressed by all compounds except gemfibrozil in two of the species, *Trachyrynchus scabrus* and *Mora moro*. 7-ethoxycoumarin-*O*-deethylase activity was depressed in *T. scabrus* by all compounds, especially diclofenac, and activity was depressed by diclofenac, galaxolide, nonylphenol and especially fluoxetine in *Alepocehalus rostratus*. Maranho et al. (2014) exposed the polychaete *Hediste diversicolor* for 14 days to sediments spiked with 0.05 to 500 ng/g carbamazepine (anti-epileptic drug), ibuprofen (analgesic), and propranolol (β-blocker drug) and 0.01 to 100 ng/g fluoxetine and 17α-ethinylestradiol (contraceptive), separately. Carbamazepine increased LP and GST activity, and GPx and AChE activity at lower concentrations only. Ibuprofen increased AChE activity and DNA damage at higher concentrations and increased LP and decreased DBF activity at intermediate concentrations. Propranolol increased EROD activity and decreased DBF and GST activity, and increased LP at lower concentrations. Fluoxetine increased EROD activity, AChE activity and LP and decreased DNA damage. 17α-ethinylestradiol increased EROD and decreased DNA damage in all but the highest concentration, and increased AChE activity and LP at lower concentrations.

Laboratory Studies: Halocarbons. Green mussels (*Perna viridis*) were exposed for 7 days to 0.1 to 1000 μg/L of either PFOS or perfluorooctanoic acid (PFOA) (Liu, Gin and Chang, 2014). Whole tissue CAT activity and hemolymph DNA damage (comet assay) increased, and relative condition factor decreased, in a dose-dependent manner for both PFOS and PFOA. Filtration rate and gill EROD activity were only decreased at 1000 μg/L PFOS, and hemolymph LMS was only decreased at 100-1000 μg/L PFOA. Liu, Chang, Gin and Viet Tung (2014) exposed green mussels to the same concentrations of PFOS and PFOA, plus perfluorononanoic acid (PFNA) and perfluorodecanoic acid (PFDA) for 7 days to examine DNA damage. All four compounds increased single strand breaks (comet assay) and chromosomal breaks (micronucleus test), and all but PFOA increased apoptosis (DNA diffusion assay), with an overall magnitude of toxicity of PFOA < PFNA < PFDA < PFOS. A 7 day depuration decreased single strand breaks in PFOS and PFNA-exposed mussels, but not chromosomal breaks or apoptosis. Rhee et al. (2014) developed an expressed sequence tags database for the Arctic green sea urchin (*Strongylocentrotus droebachiensis*) and used it to examine transcription effects of a 48 hour exposure to 1 ppb PCB (Aroclor 1254). They found that *gst-a*, *gst-m*, *gst-o*, *gst-p*, *gr*, *Mn-sod*, *Cu/Zn-sod*, *hsp70.5*, and *hsp90* were all upregulated. Fong et al. (2014) used isobaric tags for relative and absolute quantification (iTRAQ) and 2D gel electrophoresis to examine proteomic effects of dietary exposure to 2,2',4,4'-tetrabromidiphenyl ether (BDE-47) in the gonads of marine medaka (*Oryzias melastigma*). A 21-day exposure to up to 1.30 μg/g daily intake caused differential expression of up to 30 proteins in testes including down regulation of histone variants, which may disrupt spermatogenesis, and HSP90. In ovaries up to 30 proteins were also differentially expressed, including upregulation of apolipoprotein A-1 and four VTGs, perhaps indicating that BDE-47 may act as an estrogen mimic. Hong, Shen, Wang and Shi (2014) exposed marine medaka embryos to 5 to 200 μg/L technical grade hexabromocyclododecane from fertilization to 17 days post fertilization (dpf). Malformation of hatched larvae including yolk sac and pericardial edema, spinal curvature and heart malformation increased in a dose-dependent manner, along with heart rate at 5 and 8 dpf. Oxidative DNA damage (8-oxo-2'-deoxyguanosine) 6 dpf, caspase-3 activity 5 and 8 dpf, caspase-8 and -9 activity 8 dpf, *p53* and *IL-1β* expression 5 and 8 dpf and 1st fry stage, tumor necrosis factor expression 5 dpf and 1st fry stage, and apoptosis in the heart region at 1st fry stage all increased in a dose-dependent manner.

Laboratory Studies: Biocides. Chen et al. (2014) exposed marine medaka to the antifouling compounds butenolide (2.31 μg/L) and 4,5-dichloro-2-n-octyl-4-isothiazolin-3-one (DCOIT, 2.55 μg/L) for 28 days to examine proteomic effects. In male brains butenolide changed expression of 26 proteins primarily involved in DNA and RNA processing and cytoskeletal assembly, whereas DCOIT changed expression of 18 proteins primarily involved in signal transduction and cellular communication. In female brains 27 and 23 protein expressions were changed by butenolide and DCOIT, respectively, and both affected DNA and RNA processing, metabolic processes and cytoskeletal assembly similarly, with the exception of the proteins collagen alpha-1(I) chain and intermediate filament ON3-like isoform 1, which were upregulated by butenolide and downregulated by DCOIT. Gomes et al. (2014) determined that the primary form of cholinesterase in the polychaete *Capitella teleta* is AChE, and its activity had an *in vitro* 30-minute IC_{50} of 60.7 nM chlorpyrifos-oxon (active metabolite of insecticide chlorpyrifos), similar to that found in fish and molluscs. Lozano et al. (2014) exposed 3 microalgae (*Phaeodactylum tricornutum*, *Rhodomonas salina* and *Cylindrotheca closterium*) for 24 hours to Cu (5 and 10 µg/L), atrazine (herbicide, 25 and 50 µg/L) and Irgarol (antifoulant, 0.5 and 1.0 µg/L) to examine effects on antioxidant enzymes. In general, SOD and CAT activity increased and GPx and ascorbate peroxidase activity decreased, with *P. tricornutum* appearing to be the most sensitive species, and atrazine having the strongest effect. The diatom *Ceratoneis closterium* was exposed for 96 hours to nominal concentrations of Cu (2.5 µg/L), ammonia (4 mg-N/L), water-accommodated fraction of oil (1 g/L) and the herbicide simazine (300 µg/L) to examine general effects on transcription (Hook et al. 2014). The number of differentially expressed transcripts was 319, 325, 210 and 151 for Cu, ammonia, oil and simazine, respectively. Ammonia, Cu and oil downregulated most genes, whereas simazine caused a downregulation of respiration-associated transcripts, but increased the abundance of photosynthesis, purine metabolism, and stress response transcripts.

Laboratory Studies: Polycyclic Aromatic Hydrocarbons. Deng et al. (2014) developed a digital gene expression library for the testes of scallops (*Chlamys farreri*) exposed to 0.5 µg/L BaP for 10 days. Two hundred and twenty-three genes were upregulated and 828 were downregulated, including genes involved in aromatic compound catabolism, endocrine disruption, energy synthesis and immune response. European flounder (*Platichthys flesus*) exposed to contaminated sediment for 7 months showed little evidence of bioaccumulation, with only elevated muscle Pb and biliary 1-hydroxypyrene, a metabolite of pyrenes (Williams et al. 2014). There were, however, 259 transcripts and metabolites with altered abundance, which when analyzed had a similar profile to that seen in BaP exposure, and erythrocyte DNA damage (comet assay) was significantly higher than in fish exposed to reference sediment. Clams (*Venerupis philippinarum*) exposed to 2 to 50 µg/L phenanthrene for up to 15 days increased gill and digestive gland SOD activity and glutathione (GSH) content in a time and dose-dependent manner, but EROD and GST activity tended to peak at day 6 and 10 µg/L (Zhang et al. 2014). DNA damage (alkaline unwinding assay) was highest on day 3, and LP on day 6. Xia et al. (2014) exposed clams (*Ruditapes philippinarum*) to the water-soluble fraction of Shengli oil field (China) crude oil at nominal concentrations of 0.1 to 3.2 mg/L petroleum hydrocarbon for up to 15 days. Concentrations of 0.8 to 3.2 mg/L tended to cause an increase in hepatopancreas SOD, CAT, GST and peroxidase activity during the first two days of exposure, followed by a decrease back to control level (CAT) or below control levels (SOD, GST, peroxidase). Concentrations of 0.1 to 0.2 mg/L caused an increase in CAT and GST activity peaking at days 4 and 8, respectively, with no effect on SOD or peroxidase activity. GPx activity was suppressed at 0.1 to 0.2 mg/L and increased at 1.6 to 3.2 mg/L at most time points, with little effect at 0.4 to 0.8 mg/L. *Tigriopus japonicas* copepods exposed to the water-accommodated fraction of Iranian heavy crude oil for 24 hours had dose-dependent decreases in development and hatching rate and increases in GST, GR and CAT activity, with GSH content peaking at 40% and no effect on GPx activity (Han et al. 2014). Three P450 CYP3 genes were transiently upregulated: *CYP3024A2* peaking at 3 hours, *CYP3024A3* peaking at 6 hours, and *CYP3027C2* peaking at 12 hours.

Laboratory Studies: Radiation. Two bivalve species, *Pahia malbarica* and *Meretrix casta*, were exposed to a single dose of 2 to 10 Gy of γ radiation, the amount expected to be released in a nuclear accident, to examine hemocyte DNA damage via comet assay (Kumar et al. 2014). Damage increased in a dose dependent manner with little difference between species. The damage decreased over time up to 72 hours, perhaps indicating repair, but still remained significantly elevated above control.

Biomarkers of Climate Change Effects. De Souza et al. (2014) exposed Atlantic halibut (*Hippoglossus hippoglossus*) for 14 weeks to pCO_2 of 1000 µatm, equivalent to a pH drop of 0.4 from control, at both optimal temperature (12°C) and upper tolerance limit (18°C) to simulate the potential levels in 2100 in a fossil fuel-intensive scenario. Proteomic analysis indicated that at both temperatures CO_2 affected regulation of immune function proteins in the plasma, including complement component C3, fibrinogen β-chain precursor and IgM heavy chain constant region. In the gills proteins involved in energy production (*e.g.* ATP synthase, enolase-α) and cellular turnover/apoptosis (*e.g.* annexin 5 and receptor for activated protein kinase C) were affected at both temperatures, and cytoskeleton proteins (tropomyosins) were downregulated at 12°C only. Carregosa et al. (2014) exposed 2 native (*Venerupis corrugata* and *V. decussata*) and one invasive (*V. philippinarum*) clam to both low (0, 7, 14 ppt) and high (35, 42 ppt) salinities for 6 days (control levels were 21, 28 ppt). Mortality was highest for *V. corrugata*, with 100% mortality at 0, 35 and 42 ppt. In general, LP, total GSH content (all species) and GST activity (*V. corrugata*) increased at both low and high salinities, reduced GSH content and reduced/oxidized GSH ratio increased at 14 ppt only (all species), SOD activity was highest at 14 ppt and decreased at higher salinities (all species), CAT activity decreased at higher salinities (all species), and GST activity decreased at both low and high salinities (*V. decussata*). Madeira et al. (2014a) increased the water temperature from 20°C, 1°C/hour, to determine the critical thermal maximum (CTMax), the point at which loss of equilibrium occurs, for the intertidal rock goby (*Gobius paganellus*), and also sampled muscle, liver and gills every 2°C to examine effects on HSP70 and total ubiquitin levels. The CTMax was 33.1°C. Gills contained the highest levels of HSP70 and ubiquitin, and temperature increase only caused a transient increase ubiquitin at 30°C in liver. Madeira et al. (2014b) performed a similar study with juvenile gilthead seabream (*Sparus aurata*), but included intestine, hepatopancreas, muscle, gills and brain for HSP70 and ubiquitin measurements, and also measured histopathological alterations. The CTMax was 35.5°C. Transient increases in HSP70 were measured in brain (30°C), muscle and gills (34°C), and intestine (CTMax). Transient increases in ubiquitin were measured in muscle (32°C) and gills (34°C). Histological alterations from control (18°C) began to occur at 24°C in the liver and pancreatic acni, 28°C in the gills and intestine, and 34°C in muscle.

Field Methods and Pollution Indicators

Rubal et al. (2014) reported that species-rich assemblages in rocky pools (Portugal) that are subject to relatively minor pollution levels displayed taxa richness and diversity values that were not significantly different from reference pools. However, multivariate analyses revealed changes in the structure and dispersion of these assemblages when the influences of dominant taxa were down-weighted by square root transformations prior to analysis. Based on a comparison of macroalgal community composition at different sites on the Mar del Plata coast (Argentina), Becherucci et al. (2014) concluded that the order-level pattern showed a high similarity and significant correlation to the species-level pattern. Del-Pilar-Ruso et al. (2014) analyzed the responses of benthic macroinvertebrate assemblages along a pollution gradient in the western Mediterranean Sea (Spain) according to three levels of taxonomic identification: phylum, class, or order; polychaetes at family level; and syllidae (polychaetes) at genus or species level. Multivariate statistical results demonstrated the occurrence of similar distribution patterns in relation to environmental factors for the different taxonomic levels. Based on a review of five different trawl surveys from very different habitats, Brind'Amour et al. (2014) found that identification of benthic megafauna at the genus, family, and morphospecies levels provides acceptable alternatives to species-level identifications for use in community assessments. The use of morphospecies (i.e., taxa easily identified based on obvious morphological characteristics) may be especially useful for comparing community data from disparate studies using different strategies, sampling gear, and taxonomic levels.

A study of rocky shorelines located near sewage discharges and control areas on both the Atlantic and Mediterranean coasts indicated that the responses of intertidal assemblages were location dependent and that these assemblages may not serve as generally reliable indicators of any pollution effects (Cabral-Oliveira et al. 2014). In this study, the responses of macroalgae and limpets to sewage discharges were different between the Mediterranean and Atlantic coasts. Riera and de-la-Ossa-Carretero (2014) concluded that the BOPA (benthic opportunistic polychaetes and amphipods) index was generally unreliable for detecting pollution impacts in a variety of habitats in the Canary Islands. Significant differences between impact and control areas were found only in an area of harbor expansion, while weak or no effects were observed using the index in the following areas: fish cages, brine and sewage discharges, and thermal discharges. Daief et al. (2014) reported on the first application of the Multivariate AZTI Marine Biotic Index (M-AMBI) to assess the ecological status of urban sandy beaches. Use of M-AMBI enabled the identification and prioritization of urban beaches affected by sewage discharges near Casablanca, Morocco. Comparative assessments of benthic macroinvertebrate assemblages in coastal areas off the Yangtze River Estuary using M-AMBI indicated that the ecological status was seriously degraded, while use of an alternative index (AMBI) indicated only a slight degradation (Liu, Li, Lin, Cai and Wang 2014). M-AMBI was recommended for use in Chinese coastal areas because of the integration of diversity and species richness into the index. A study of benthic assemblages in Victoria Harbor (Hong Kong) demonstrated that benthic macrofauna and free-living nematodes respond differently to a gradient in organic enrichment of the sediments (Xu, Cheung and Shin 2014). As a group, the macrofauna appeared to be more tolerant of the pollutant stress, and the use of both biotic groups was recommended for understanding the overall response of the benthic ecosystem. Niquil et al. (2014) concluded that none of the classical trophic-dynamic concepts tested in a model application (food-chain length, detritivore/herbivore ratio, and trophic efficiency) are reliable as operational metrics to evaluate the impacts on the trophic state of an ecosystem.

Pawlowski et al. (2014) identified gradients of foraminiferal assemblages near two salmon net cage facilities on the west coast of Scotland and tested the applicability of next-generation sequencing (NGS) of DNA and RNA extracted from sediment samples. The results indicated that NGS metabarcoding of foraminiferal species has the potential to be a valuable tool for assessing impacts to sediment assemblages of forams in the marine environment. A measured series of differentially expressed genes were recommended as potential biomarkers of polycyclic aromatic hydrocarbon (PAH) exposure in the commercially important clam *Venerupis philippinarum* (Liu, Pan, Gong, Tao, Hu and Miao, 2014). The modulation of the expression of these genes from PAH exposure may affect immune defense of the clam and may have a causal relationship with observed mass mortalities of the species. Analyses of stable nitrogen isotopes in the limpet *Patella caerulea* in the northern Adriatic Sea indicated that this information provides a useful indicator of organic matter uptake from anthropogenic sources (Rožič et al. 2014). The $\delta^{15}N$ in limpets displayed only small differences according to the size of sampled organisms, and variations between tissues for individual organisms were less than 1 percent. Evaluation of $\delta^{15}N$ for various marine biota of the Glician Coast (Spain) showed that conclusions concerning the contribution of anthropogenic sources were highly dependent on assumptions used to model contributions of alternative nitrogen sources (Bode et al. 2014). Based on these results, the authors concluded that $\delta^{15}N$ levels alone do not provide a reliable assessment of the influence of anthropogenic nitrogen in coastal waters. Martins et al. (2014) used a variety of fecal sterol measurements and associated molecular marker indices to describe the extent of sewage pollution in Babitonga Bay (Brazil). Comparisons of the sediment sterol concentrations with sterol index limits in the current literature suggested that such limits may underestimate the magnitude of sewage contamination in the subtropical Bay. Analytical techniques were described for the quantification of five pharmaceutical residues in wastewater effluents, receiving waters, and caged mussels off the Irish coast (McEneff et al. 2014). Three of the five pharmaceuticals were detected in mussel tissue at concentrations of 4–29 ng g^{-1} dry weight. The results indicated the value of *Mytilus* spp. as bioindicators for this class of substances in the marine environment. A survey of urea levels in the Gulf of Trieste demonstrated that urea is a reliable indicator of the extent of sewage discharges in the coastal zone (Cozzi et al. 2014). Urea was found to be more sensitive and specific than other nutrients, and it is important in biogeochemical nitrogen cycles in the Gulf.

The Catlin Seaview Survey is a new framework for monitoring of coral reefs using underwater imagery (González-Rivero et al. 2014). The computerized framework maps a variety of coral reef characteristics with a spatial resolution of 2 to 6 m^2. Rengstorf et al. (2014) reviewed the methods for predicting the distribution of benthic biota in the deep sea and evaluated the use of several kinds of information for predicting the occurrence of the deep-water coral *Lophelia pertusa*. The results demonstrated that predictive models incorporating vertical and horizontal demersal flow patterns are substantially better than models using only substrate variables in predicting *Lophelia* distribution. In the Learmonth Bank area of northern British Columbia, Neves et al. (2014) used multibeam sonar (backscatter, bathymetry, and slope) to characterize bottom substrates. Subsequent analyses and comparisons with ROV surveys demonstrated significant correspondence between substrates and coral/sponge biotopes, indicating that substrate characterizations are useful surrogates in predicting biotope distributions for this area. High-resolution sidescan sonar (410 kHz) and multibeam echosounder data were used to successfully produce high-precision maps of reefs of the tubiculous polychaete *Sabellaria spinulosa* near an offshore windfarm in the North Sea (Pearce et al. 2014). Post-construction maps of the site revealed that the wind farm was having a positive influence on the overall biodiversity of nearby areas. Reshitnyk et al. (2014) compared the applicability of high-resolution satellite imagery and a 200-kHz acoustic system for mapping the distribution of submerged macrophytes in British Columbia (Canada). The satellite imagery was accurate at depths less than 3 m and was able to distinguish various macrophyte types and unvegetated substrate. The acoustic technique was more accurate for the distribution of eelgrass but was unable to distinguish between some algal types. Beeden et al. (2014) described a participatory monitoring program for the Great Barrier Reef (GBR) that provides current information to resource managers on benthic community composition, reef condition and impacts, coral diseases, and presence of debris. The program has been refined over a 10-year period and accumulates observational data from rangers, tourism operators, and the general public. Although specific to the GBR, the protocol could be modified to meet the needs of resource managers at other reef areas.

Based on an assessment in Rijeka Harbor (Croatia), Frančišković-Billinski and Cukrov (2014) concluded that analysis of bulk sediments (<2 mm) is a better approach to assess contamination of marine sediments than analysis of the fine fraction (<63 µm). No significant statistical differences between the sediment fractions were detected for 19 elements using multivariate analyses and comparisons of element concentration ratios. A group of abiotic variables was found to explain about 42 % of the variability in rocky reef community characteristics on the Queensland coast (Australia) (Richmond and Stevens 2014). The correlation with purely physical characteristics was relatively poor (explaining 22 % of variation), but the predictive ability of the statistical model increased dramatically when fishing pressure was included in the combination of explanatory factors. Foster et al. (2014) presented guidelines for designing surveys of benthic habitats using autonomous underwater vehicles (AUVs). In cases where there is no *a priori* information on the distribution of environmental covariates, structured 2-dimensional designs (grid, stratified, or GRTS) are recommended. Kernal density estimation surfaces were demonstrated as a novel and reliable technique for interpretation of trawl data in the Grand Bank/Flemish Cap area of the northwest Atlantic (Kenchington et al. 2014). When used in conjunction with literature data, this approach can be useful for defining vulnerable marine ecosystems, such as assemblages of large sponges, sea pens, and gorgonian corals in the study area.

Effects of Wastewater Discharges

Díez et al. (2014) described a long-term (1984–2012) monitoring program of rocky subtidal macrophytes along the Basque coast (Spain) during a period of sewage treatment upgrades for a large metropolitan area. Although some recovery of macrophyte assemblages was observed, the development of canopy-forming macrophytes that are characteristic of reference areas had not occurred during a period up to 11 years after implementation of secondary sewage treatment. Rocky shore macroinvertebrate assemblages near a sewage discharge on the Portuguese coast had higher secondary production levels than reference areas (Cabral-Oliveira, Dolberth and Pardal (2014). The increase in production resulted from higher abundances of tolerant suspension-feed biota near the discharge. The long-term decline of marine algal taxa near Igidea (Korea) was attributed to a nearby sewage discharge (Shin et al. 2014). Ganesh et al. (2014) characterized a 3- to 6-km gradient of benthic macroinvertebrate assemblages offshore of a sewage discharge in the surf zone of the Bay of Bengal (India). Although the outfall is located in a high-energy environment, the nearby benthic communities were dominated by opportunistic spionid polychaetes. A before-and-after study of a new offshore sewage discharge in the Brać Channel (Adriatic Sea) revealed an increase in phytoplankton biomass in the deep zone after initiation of the discharge, but there was no associated phytoplankton bloom or oxygen depletion (Skejić et al. 2014). In the post-discharge period, there was an increase in phytoplankton species diversity, indicating that moderate nutrient inputs do not necessarily result in adverse effects. Bănaru et al. (2014) described the variability of $\delta^{15}N$ values according to zooplankton size and seasons in the Bay of Marseille. The study indicated that contaminated particulates from local sewage discharges are being mixed with marine phytoplankton (mainly pico- and nanoplankton) and are transferred up the zooplankton trophic links.

Berreta et al. (2014) described the levels of various pharmaceutical and personal care products (PPCPs) in sediments in a bay in Brazil that has received domestic and hospital sewage for extended periods. The highest measured concentrations were associated with the fragrances galaxolide (52.5 ng g^{-1}) and tonalide (27.9 ng g^{-1}), with pharmaceuticals being measured at lower concentrations of 0.7 to 14.3 ng g^{-1}. The endocrine-disrupting chemicals 4-nonylphenol (NP) and bisphenol A (BPA) were detected in seawater samples from the Cape D'Aguilar Marine Reserve (Hong Kong) at mean concentrations ranging from 62.5 to 392.5 ng L^{-1} (Xu et al. 2014). Based on a risk quotient approach, the ambient concentrations of NP were predicted to pose a potential hazard to marine organisms. A comprehensive survey of polar organic micropollutants in nearshore seawater samples from a variety of global locations revealed that caffeine was the most frequently detected compound, at concentrations up to 3068 ng L^{-1} (Nödler et al. 2014). The high detection frequency of caffeine in these samples (95.4 percent) indicates the widespread occurrence of untreated sewage discharges in coastal areas.

Effects of Dredging and Extraction

Hwang, Lee, Choi et al. (2014) documented negative effects on fish assemblages of sand extraction activities in Gyeonggi Bay (Korea). When compared with two non-mining areas, the sand mining area had significantly lower values for total fish abundance, species richness (R1), and species diversity (H'). Based on a review of marine surveys at seven sand extraction sites off Korea, Kim et al. (2014) concluded that turbidity plumes may be transported considerable distances from the extraction site and have adverse effects on the marine environment. Recommendations are presented for monitoring strategies and for minimizing adverse effects of sand extraction operations. Monitoring of a dredged material disposal site in the Saronikos Gulf (Greece) using sediment toxicity tests (Microtox®) and various biomarkers in caged mussels indicated contaminant effects primarily associated with high levels of PAH and AH (Tsangaris et al. 2014). Elevated hydrocarbons and associated sublethal effects in caged mussels were not persistent after cessation of disposal activities, indicating that effects on water column organisms are transient. Alternatively, sediment contamination and toxicity persisted after termination of disposal. De Backer et al. (2014) observed similar changes in benthic macroinvertebrate assemblages in the North Sea as the result of three different kinds of human activities: sand extraction, dredged material disposal, and construction of offshore wind facilities. All three activities resulted in a change in sediments toward finer particle sizes, which led to increases in macrobenthic species typical of muddy sands. The overall effect on community structure was an increase in biodiversity for all three sites.

Field Survey Assessments

Sediment concentrations of various organic substances near the port areas of Athens (Greece) indicate that this region is one of the most contaminated sites in the Mediterranean (Kapsimalis et al. 2014). Maximum aromatic hydrocarbon concentrations were dominated by high-molecular-weight PAHs and ranged up to 4,457 µg g^{-1}. An analysis of the chemical inputs and impacts of a submarine groundwater discharge (SGD) in the Ionian Sea indicated that the water column at the site had moderate to bad quality, based on a eutrophication index (Pavlidou et al. 2014). However, the SGD has insignificant adverse effects on local benthic macroinvertebrate assemblages. Stewart et al. (2014) reported that the concentrations of a wide variety of emerging contaminants (e.g., flame retardants, pharmaceuticals, and alkylphenols) in sediments near Aukland (New Zealand) were similar to those reported for worldwide studies. Multifactor models were reasonably accurate in predicting uptake of PCBs and polybrominated diphenyl ethers (PBDEs) in benthic macrofauna in the Strait of Georgia (Canada) (Burd et al. 2014). Overall, PBDEs were accumulated more extensively than PCBs, indicating that PBDE transfer to benthos is more dependent on recent organic detritus and associated contaminant levels from ongoing discharges. Slijkerman et al. (2014) documented some evidence of eutrophication based on nitrogen concentrations recorded near the island of Bonaire, which has some of the highest quality coral reef habitats in the Caribbean. A group of liver and gonadal abnormalities were identified in large-toothed flounder (*Pseudorhombus arsius*) and oriental sole (*Synaptura orientalis*) that may provide useful biomarkers for assessment of contaminant effects in Kuwait Bay (Stentiford et al. 2014). The highest prevalences were measured for foci of cellular alteration (FCA) in the liver and putative intersex (ovotestis) in the gonads of both species.

Hard-bottom assemblages near a discharge of iron process mine tailings in northern Chile were dominated by encrusting species that had replaced the normal macroalgae that inhabit nearby control areas (González et al. 2014). Overall species diversity near the discharge was similar to diversity at control locations, indicating that the observed effects were caused by species replacement, rather than a decrease in overall species richness. Intertidal areas of Baynes Sound (British Columbia, Canada) used for intensive aquaculture of clams were found to have higher abundances of the invasive species *Batillaria* sp. and *Hemigrapsus oregonensis* (Bendell 2014). The aquaculture areas are seeded with juvenile clams and appear to attract the important predatory crab *H. oregonensis*, with unknown effects on overall ecosystem functioning. Mirto et al. (2014) found that nematode species composition in areas near fish farms in the Mediterranean Sea were significantly different from reference areas, but abundance, biomass, and biodiversity did not change near the aquaculture facilities. The nematode genera *Richtersia, Desmoscolex*, and *Halalaimus* were very sensitive to discharges from fish farms and almost completely disappeared from the nearby sediments. Benthic macrofaunal assemblages at two Antarctic stations (McMurdo and Casey) had similar relationships with sediment contamination for community composition and biodiversity (Stark et al. 2014). However, very different responses to contaminated sediments were observed for crustacea (primarily amphipods), which decreased at McMurdo and increased at Casey.

The first scientific survey of fish and invertebrate communities on oil platforms off West Africa documented the presence of diverse platform assemblages that are very different from the soft-bottom shelf communities that are predominant in these waters (Friedlander et al. 2014). The total estimated fish biomass for some individual platforms exceeded 1 ton and was dominated by barracuda (*Sphyraena* spp.), jacks (carangidae), and rainbow runners (*Elagatis bipinnulata*). Following a period in 2010 of warm water and increased storms and runoff, the shallow-water reefs in St. John (U.S. Virgin Islands) showed relatively minor adverse effects in relation to the magnitude of potential stresses (Edmunds and Gray 2014). The detected effects of the unusual year were detected in 2011, in the form of decreases in coral recruitment and higher densities of serpulid polychaetes on settlement tiles placed near the reefs.

Resident male and female bottlenose dolphins in Doubtful Sound (New Zealand) displayed different foraging responses to human disturbance by tour vessels (Symons et al. 2014). When boats were present, male dolphins increased bottom time and conducted fewer dives, while females decreased bottom time and performed more dives. Based on comparisons with an energetic model, it was concluded that both sexes experienced decreased net energy gains because of vessel interactions. Smith et al. (2014) analyzed the impacts of large, seasonal hypoxia events in the Gulf of Mexico on the fishery for brown shrimp (*Farfantepenaeus aztecus*). Empirical fishery data for the Gulf were generally consistent with the model predictions. A review of environmental considerations for offshore geological storage of CO_2 indicated that leakage from such facilities can have substantial effects on sediments, biogeochemical cycles, and marine biota (Carroll et al. 2014). Recommendations are presented for development of baseline surveys at candidate sites. Lange and Griffiths (2014) presented the results of a large-scale survey of epibenthic megainvertebrates off the west coast of South Africa that may be useful in the future designation of marine protected areas.

Effects of Fishing Activities

Grabowski et al. (2014) reviewed available literature to develop a framework to assess benthic impacts of the various fixed and mobile fishing gear common to New England waters. In general, both susceptibility and recovery scores were highest for hydraulic dredges, intermediate for otter trawls and scallop dredges, and lowest for fixed gear. When compared with a fishery closure area, a demersal trawl fishing area on the continental shelf in the Tyrrhenian Sea (Sicily and Calabria, Italy) displayed a lower density index (DI) and number of species (S) for infaunal communities (Mangano et al. 2014). The fished sites also had an absence of characteristic burrowing decapod species, and the assemblages were dominated by subsurface deposit feeders and opportunistic polychaete taxa. Muntadas et al. (2014) described the changes in benthic community structure and productivity at a trawl site in the northwestern

Mediterranean Sea when compared with a non-fished area. The community changes in the trawled area may be beneficial to adults of the commercially important red mullet, but may cause adverse effects for red mullet recruits. This study demonstrated that permanent fishing closures that allow recovery of benthic communities may more beneficial to demersal fish species than temporary closures. Van Denderen et al. (2014) found a negative relationship between trawling intensity and species richness of benthic communities in the North Sea. The results of this study emphasized the importance of the spatial scale used to assess trawl impacts because of complex relationships among factors associated with habitat heterogeneity, trawl intensity, benthic community structure, and community productivity. Studies of trophic position ($\delta^{15}N$) and energy flow in benthic communities in the North Sea demonstrated that either species- or community-level energy flows are good indicators to detect changes in benthic community function after cessation of trawling for 14 months (Dannheim et al. 2014). Because of the cessation of trawling activities in areas developed as future wind farms in the North Sea, there will be an excellent opportunity to monitor the recovery of structural and functional characteristics of benthic communities.

Comparisons of a trawling and dredge fishing area to a nearby fishery exclusion zone in New Zealand indicated that the substrate conditions had been highly modified by the fishing activities (Handley et al. 2014). The baseline demersal habitat was characterized by a shell-gravel surficial layer and the presence of large molluscs, while the fished areas had a homogeneous fine mud bottom with reductions of species richness and large-bodied taxa. Hydraulic dredging for cockles in Dundalk Bay (Ireland) was found to have no significant effect on sediment characteristics or benthic community structure (Clarke and Tully 2014). The dredging operations resulted in a relatively short-term decrease in densities of the abundant bivalve *Angulus tenuis*. Meseck et al. (2014) reported that a one-time dredging for northern quahog clams (*Mercenaria mercenaria*) in Long Island Sound had only minor effects on a wide variety of sediment chemical characteristics. Short-term differences in organic nitrogen and hydrogen flux were measured following dredging, but these effects disappeared within weeks. Experimental scallop dredging in the Firth of Lorn (Scotland) resulted in significant changes in epifaunal community composition at two of the three study sites (Boulcott et al. 2014). Based on analyses of the effects of dredging over different substrates, it was concluded that pebble and cobble substrates could be especially susceptible to dredging effects on epifauna. A series of photographic surveys of deep-water benthic habitat and megafauna at three potential marine protected areas (MPAs) on the Hebridean Slope (Scotland) revealed the presence of trawl marks in all areas (Hughes 2014). Trawl marks were observed over the complete depth range, but peak abundances were observed in the north area at depths of 1,300 to 1,400 m.

A survey of bait collectors on the sandflats of Langebaan Lagoon (South Africa) revealed that large amounts of sediments were disturbed (6,189 tons), and large areas were trampled (>500,000 m^2) (Nel and Branch 2014). Although the harvesting of sandprawns for bait did not constitute a threat to the stock, the sediment disturbances resulting from these activities were predicted to have adverse effects on the lagoon ecosystem. Long-term harvesting of intertidal clams was found to have measureable effects on the small-scale distribution patterns of meiofaunal nematodes (Boldina et al. 2014). Clam digging resulted in an attenuation of the normal aggregation patterns of nematodes and a lack of a cyclical pattern in the spatial variogram model. Purroy et al. (2014) developed an interactive map of a large marine protective area in northeast Spain that displayed cumulative impacts of four fishing-gear types: bottom and surface longlines, trammel nets, and gill nets. The greatest potential impacts were associated with areas near the coast and along canyon margins. Surveys of four deep rocky habitats in the Tyrrhenian Sea documented extensive damage to coral colonies from lost fishing gear, consisting primarily of long lines (Bo et al. 2014). Clear injuries to hard-bottom communities were observed in 19 to 62 % of the video frames analyzed from the survey. In a mesocosm experiment, Ingels et al. (2014) assessed the potential indirect effects of trawling disturbances by exposing natural nematode communities in soft sediments to varying densities of large macrofauna that may be reduced by trawling. Removal of large bioturbating macrofauna (e.g., ophiuroids, polychaetes, and bivalves) resulted in indirect effects on nematode community structure. Based on an individual-based model of harbor porpoises in Danish waters, Nabe-Nielsen et al. (2014) estimated that annual by-catch rates from fisheries ≥10 % would result in a monotonic population decline that ultimately leads to extinction of the local population. For the 1990s, the estimated by-catch in the Danish North Sea was approximately 4.1 % of the porpoise population, resulting in a predicted substantial population reduction for the species.

Effects of Marine Debris

Marine debris studies continue to address gaps in understanding about the impacts of all types of debris, especially derelict fishing gear and consumer plastics.

Reviews and Overviews Ivar do Sul and Costa (2014) analyzed 101 publications on microplastic debris and found 60% were written since 2009. Lavender Law and Thompson (2014) provides a brief overview of the state of microplastic research and a perspective on future needs including a better understanding of how often marine organisms encounter microplastic fragments in the natural environment and the interactive risks posed by such encounters. A three-pronged summary of the issues posed by microplastics in the marine environment was published in Koelmans et al. (2014b), with separate and intersecting perspectives from the pharmaceutical industry, academia, and government which all discussed potential solutions for improved education, recycling, and waste management. In agreement with the concept of multi-stakeholder engagement, Lee et al. (2014) found success in addressing the impacts caused by polystyrene buoys in South Korea by implementing a series of participatory workshops to develop policy solutions, such as subsidizing removal of polystyrene buoys after their effective lifecycles to minimize polystyrene fragmentation in marine systems.

Effects on Tourism. Jang et al. (2014), estimated the economic impacts to tourism caused by a marine debris event driven by excessive rainfall on Geoje Island, South Korea. Authors considered the decreased number of visitors and typical expenditures when developing an estimate of lost tourism in the amount of $29-37 million U.S. dollars (Jang et al. 2014), and suggest countermeasures such as cost sharing by local governments as implementation of the "polluter pays" principle and management of river dams that could be employed to trap debris before it flows downstream. An approach to surveying recreational users of rocky shores in the UK and internationally, in order to determine perceptions of recreational uses and their effects on the environment, spontaneously identified littering as a concern of coastal users as it was a commonly mentioned theme in open-ended questions about recreator insights about factors that affect personal enjoyment of coastal areas (Wyles et al. 2014). This finding suggests marine litter impacts both the ecological environment, as noted in the remainder of this literature review, as well as the social environment through negative impacts to user experiences during recreational trips to coasts (Wyles et al. 2014).

Global Scale Impacts and Directives. A literature review examined cetacean interactions with marine debris on a global scale, determining that the aggregated published literature reports 48 cetacean species (56%) have ingested marine debris (Baulch and Perry 2014). In Europe, scientists supporting the Marine Strategy Framework Directive reviewed potential suitability of marine vertebrates as species to include in their long-term debris monitoring studies (Galgani et al. 2014) and recommended sea turtles as useful species given known ingestion rates and population status. This recommendation will likely drive future research and monitoring of marine litter in European waters. For instance, sea turtle exposure to marine debris in the Atlantic is analyzed by Gonzalez Carman et al. (2014).

Entanglement in Marine Debris. Given the difficulties in surveying animals that have become entangled in marine debris in the wild, it is not a simple task to determine the prevalence of entanglement for marine species and the effect of such entanglements on marine populations. Generally, the only records come from organisms with severe morbidity or mortality that strand on coasts or are pulled to the surface when derelict fishing nets are retrieved. Two reviews were published in 2014 that assess the state of knowledge on entanglement of cetaceans, pinnipeds, sea turtles, seabirds, and other marine species (NOAA MDP 2014a, WAP 2014). Reviews aggregated records showing that in the United States, 44 seabird species, 9 cetacean species, 11 pinnipeds, 31 invertebrates, and all 6 sea turtle species present in the U.S. have been reported entangled in various types of marine debris. Rates of entanglement appear greater where human populations overlap with convergence zones of high fishing pressure or debris accumulation (NOAA MDP 2014a). In WAP (2014), authors documented country-specific case studies in Australia, the United Kingdom, and the United States and Canada, and used these as a springboard to suggest a global ghost gear initiative to drive solutions in ghost gear hotspots, promote learning from valuable case studies, and enable global monitoring to better understand the issues and solutions.

Interactions with Derelict Fishing Gear (DFG). Ghost fishing is the indiscriminate take of marine organisms in derelict fishing gear, and may affect target or non-target species of the fishery. A rarely discussed impact of derelict fishing gear is smothering of important marine and coastal habitat. For example, research in the Florida Keys showed submerged marine debris is mainly sourced from the commercial spiny lobster fishery, and was not aggregated in areas of high fishing effort but instead tended to aggregate in sensitive habitat such as coral reefs due to wind and storm action. This over-represented the damaging impacts of derelict traps in coral habitat, with each trap causing damage over a 0.6 m^2 footprint from movement, breakage, and smothering (Uhrin et al. 2014). Through the use of remotely operated vehicles to survey rocky banks in the Mediterranean between 70-280 m deep, Bo et al. (2014) demonstrate fishing gear impacts such as broken coral colonies and scattered habitat-forming species. This study demonstrated widespread presence of various forms of marine debris, especially long lines, trawl nets, other fishing-associated gear, and litter, and establishes a relationship between presence of debris and decreased coral community health (Bo et al. 2014).

In Anderson and Alford (2014), ghost fishing in Louisiana was examined through citizen science efforts to remove derelict blue crab traps. Of the 3,607 traps collected during removal events, more than 65% were actively ghost fishing and a total of nineteen different species were identified as either captured or killed within traps (Anderson and Alford 2014). In the Chesapeake Bay, Bilkovic et al. (2014) continued a research program investigating the spatial pattern of derelict blue crab traps in aggregating areas or hotspots. Bilkovic et al. (2014) documented through field research 31,546 marine organisms and 40 species captured in retrieved traps, and used these data to estimate over 900,000 crabs are killed in derelict blue crab traps each year. The majority of non-target species trapped and killed in blue crab traps each year are demersal species (>95%) including Atlantic croaker (*Micropogonias undulates*), black sea bass (*Centropristis striata*), white perch (*Morone americana*), and catfish species (*Ictaluridae sp*). A total of 46 diamondback terrapins (*Malaclemys terrapin*) were captured and subsequently drowned in derelict crab traps (Bilkovic et al. 2014). In contrast to the Chesapeake Bay, fish traps in the Caribbean are used to catch a variety of species. Field experiments investigated simulated derelict traps and tracked associated fish assemblages, fish behavior, and mortality over six months (Renchen et al. 2014). Authors determined that escape panels did not function as intended, but compared to other locations such a the Chesapeake Bay and Gulf of Mexico, a small number of fish mortalities were reported (2% trapped fish) (Renchen et al. 2014). In Alaskan waters near Womens Bay, Kodiak, 192 red king crabs (*Paralithodes camtschaticus*) were tracked from 1991 to 2008 (Long et al. 2014). Thirteen tagged crabs were killed in derelict fishing gear, while another twenty were captured in ghost fishing traps and released by divers. Using these parameters and life history, population, and fishery harvest data, authors estimate 16-37% of the population with carapace length greater than 60 mm were killed during the study (Long et al. 2014). This represents a significant portion of the fishery and may indicate a situation in which incidental mortality from ghost fishing should be considered in stock assessment models.

Arthur and Sutton-Grier et al. (2014) published a synthesis of several derelict fishing trap studies across the United States, highlighting some of the similarities among various trap fisheries in terms of the number of derelict traps present, the percentage of derelict traps that ghost fish, and the overall impact of traps on the number of target organisms captured or killed per year. Regional differences were important lines of evidence in showing the dynamics of derelict fishing gear interactions are driven by local factors. Authors present options for a derelict fishing trap management strategy that targets policy options to educate the public and fishermen about ghost fishing, and to develop more effective gear options to minimize loss and ghost fishing given the persistence of traps and the fact that traps did not perform as regulated.

Unintended consequences between marine organisms and fishing gear that has been lost or discarded are difficult to document given the wide expanse of marine waters affected by derelict fishing gear. Documenting interactions between fishing gear and marine species is important, however, when considering how to reduce such occurrences and maintain healthy populations of fishery species, seabirds, turtles, and marine mammals. Yorio et al. (2014) examine a kelp gull (*Larus dominicanus*) colony in which 27 adults were entangled by monofilament fishing lines used by recreational fishers. Authors suggest that foraging behavior along the coastline may lead to entanglement in lines, which are then brought to the colony and cause severe morbidity and mortality when the line becomes entangled in vegetation (Yorio et al. 2014). A study by Adimey et al. (2014) summarize derelict fishing gear interactions with bottlenose dolphins (*Tursiops truncatus*), Florida manatees (*Trichechus manatus*), and loggerhead (*Caretta caretta*), green (*Chelonia mydas*), leatherback (*Dermochelys coriacea*), hawksbill (*Eretmochelys imbricata*), Kemp's ridley (*Lepidochelys kempii*), and olive ridley (*Lepidochelys olivacea*) sea turtles that stranded in Florida between 1997 and 2009. Overwhelmingly, cases were dominated by hook and line interactions with monofilament fishing line (75.3%), and regression analysis showed interactions with marine debris increased over time for dolphins, manatees, loggerhead sea turtles, and green sea turtles (Adimey et al. 2014). Barreiros and Raykov (2014) report three recent cases of loggerhead sea turtle interactions with fishing gear, in which one turtle swallowed long line and several hooks and was subsequently euthanized due to the damage this caused; one turtle had a limb amputated due to a fragment of tangled nylon long line; and one turtle was enmeshed in a tangle of fishing line such that it caused a limb to atrophy and amputate in the wild. It is suggested that occurrences of fishing gear interactions with marine organisms are more common than scientists are aware.

The literature also contains a number of reports and short communications such as the one by Barreiros and Guerreiro (2014) which provides notes on a bream

(*Pagellus acarne*) entangled in a hollow plastic bottle lid. The fish was caught by hand near the Azores and rehabilitated after removing the lid, which was acting as a collar. Aggregating these reports to understand the frequency and nature of species interactions with marine debris will be important to focusing future solutions-based research efforts.

Debris Ingestion-Invertebrates. Direct ingestion of debris has a suite of potential harmful effects, including dietary dilution, gut blockage, and starvation; laceration and secondary infection; and potentially more subtle effects on hormone function, reproduction, and the immune system that are compounded by the suite of chemicals that may associate with plastic debris as it is exposed to waters and sediments contaminated with substances such as PCBs, PAHs, trace metals, and chemicals integral to the plastic resin itself (NOAA MDP 2014b). In a review of the incidences and health effects of ingesting marine debris, authors at NOAA discuss that marine debris may be ingested directly or consumed indirectly though contaminated prey (NOAA MDP 2014b). The distinction is not novel, but indirect consumption has not been an active line of research to date and is one of many potential ways to consider the "source" of debris in terms of the pathway from point of loss to environmental effect. This section on ingestion of marine debris shows the overwhelming focus of microplastic ingestion studies on invertebrate organisms that form the base of many food webs. These studies lay groundwork for future research evaluating indirect ingestion of debris and the issues that may arise from higher trophic-level organisms eating contaminated prey. For a detailed review of the species-specific impacts of debris ingestion, see NOAA MDP 2014b.

Studies examining the ingestion of microplastics by marine invertebrates cover zooplankton (Frias et al. 2014; Setala et al. 2014), amphipods (Chua et al. 2014), isopods (Haemer et al. 2014), sea urchins (Della Torre et al. 2014; Kaposi et al. 2014), bivalves (Van Cauwenberghe and Janssen 2014), and crabs (Watts et al. 2014). Frias et al. (2014) conducted a study sampling Portuguese waters for microplastics associated in surface waters with plankton, in which 61% of samples (n=93) contained microplastic particles. This study is mentioned here as a placeholder for the ongoing monitoring campaigns and studies that are determining environmental concentrations of marine debris and in particular microplastic debris in the oceans. Concentration and density of particles are important variables in risk assessment approaches and fluxes should be carefully considered.

Chua et al. (2014) examine marine amphipod (*Allorchestes compressa*) assimilation of microplastic particles that were isolated from facial soap, and added complexity to the experiment by additionally exposing the amphipods to PBDE congeners. Results were somewhat inconsistent, showing reduced PBDE uptake in the presence of microplastic, but greater (proportional) uptake of the higher brominated congeners (e.g., BDE-153 and BDE-154) (Chua et al., 2014). The marine isopod (*Idotea emarginata*), when offered a food source embedded with fluorescent microplastic particles, consumed the food source without microplastic content (Haemer et al. 2014).

Fecal analysis coupled with fluorescent tracing through the digestive tract confirmed the same concentration of particles were ingested and egested, and accumulation or translocation to tissues did not occur (Haemer et al. 2014).

Sea urchin (*Paracentrotus lividus*) embryotoxicity was considered in Della Torre et al. (2014) through laboratory experiments with nano-sized polystyrene of two surface charges determined by functional groups (carboxylated and amine). The amine-group polystyrene nanoparticles caused developmental effects at 3.85 ug/mL (EC50 at 24 hours post-fertilization) and 2.61 up/mL (EC50 at 48 hours post-fertilization), which may be in part due to the greater dispersion of these particles in the seawater medium. However, in Kaposi et al. 2014, sea urchins (*Tripneustes gratilla*) at the larval stage ingested polyethylene spheres at rates proportional to the treatment concentration, though environmentally relevant concentrations did not have a determinable effect on larval growth. Watts et al. (2014) determined in a laboratory experiment that the shore crab (*Carcinus maenas*) uptake of microplastic particles may occur through inspiration in the gills or through uptake in exposed food. Ingested plastic was retained in tissues for 21 days following inspiration and 14 days following ingestion, indicating the exposure route may influence the physiology involved with trapping and excreting foreign microplastic particles (Watts et al. 2014).

In an effort to focus research on organisms often consumed by humans, Van Cauwenberghe and Janssen (2014) purchase oysters (Crassostrea gigas) and blue mussels (Mytilus edulis) intended for consumption and analyzed edible tissues for microplastic concentration. Methods allowed for the full depuration of the organism before tissue analysis, which combined showed an average of 0.36 +/- 0.07 particles per gram (wet weight) in mussels and 0.47 +/- 0.16 particles per gram in oysters. Authors translate this to human ingestion of approximately 1,800-11,000 microplastic particles per year based on a range of mussel and oyster dietary exposures, but caution extrapolation to human health risk assessment (Van Cauwenberghe and Janssen 2014). The implications of marine debris as a source and pathway of chemicals into marine food webs, in terms of human consumption of marine fish and invertebrates, are yet to be determined but will need to be understood within the context of accepted routes of chemical exposure.

Debris Ingestion: Birds*.* Avian species were perhaps some of the first to be studied for marine debris ingestion, and numerous species have been documented to ingest plastic fragments as well as fishing gear and line. In Acampora et al. (2014), scientists compare juvenile and adult ingestion rates in short-tailed shearwater (*Puffinus tenuirostris*) carcasses obtained through stranding events. No significant relationship was determined between incidence of ingested debris and overall body condition, though perhaps concerning was the finding that juvenile birds had a higher frequency of ingestion and a higher ingestion concentration than adults. Bond et al. (2014) investigated stranded seabird species near Sable Island, Nova Scotia, for ingested debris and noted >72% occurrence in northern fulmars, sooty shearwaters, and great shearwaters. Lavers et al. (2014) sampled feathers and stomach contents from flesh-footed shearwaters in Australia, and determined a significant relationship between increasing debris body burdens and reduced body condition and contaminant load. Many shearwater chicks obtained debris during parental feeding, and fledglings could attain significant amounts (ranging from 0.13-3.21 grams) of plastic during a single feeding which insinuates this as a major acute route of exposure (Lavers et al. 2014). Within seabird communities, it has been documented that certain species will use debris items within their nest structures. Verlis et al. (2014) document this behavior in brown booby (*Sula leucogaster*) nesting near the Great Barrier Reef, Australia. Authors surveyed 96 nests and found 58.3% contained marine debris, with an average of four pieces per nest structure and a predominance of hard plastic items versus other forms of debris such as fishing line.

Donnelly-Greenan et al. (2014) investigated prey abundance in relation to plastic ingestion in northern fulmars (*Fulmarus glacialis*) near Monterey Bay, California, and noted a significant increase in the plastic categories found in birds stranded from 2003-2007 though no direct relationship with body condition was noted. A broad study of the incidence of plastic accumulation in North Atlantic marine birds determined the highest prevalence of ingested plastic in great shearwaters (71%) and northern fulmars (51%), and in the case of the great shearwater, the highest concentration of plastic fragments in the digestive tract. These broader baseline studies are useful in evaluating trends over time and point to the potential utility of large-scale data aggregation systems and databases. In a review focused on ingestion of marine debris by seabirds in Canada, Provencher et al. (2014b) found debris ingestion data for only 6 of 91 total seabird species inhabiting Canadian waters, and with additional reports without data for 33 species. Authors propose a focus on characterizing the risk of plastic fragment ingestion and continued monitoring of potential indicator species (Provencher et al. 2014b).

Debris Ingestion: Turtles. Several studies documented the incidence, frequency, and type of plastic debris ingested by sea turtles, including Camedda et al. (2014) which focused on loggerhead sea turtles in Sardinia and determined higher prevalence of user plastics, sheets, and fragments in this population. Hoarau et al. (2014) provided one of the first records of debris ingestion in Indian Ocean loggerhead sea turtles, noting debris in 51% of gut or fecal samples but not finding a relationship between the gut contents of dead individuals and the fecal contents of live turtles. In two peer-reviewed publications and a dissertation published in 2014, Schuyler analyzed global patterns of anthropogenic debris ingestion by sea turtles (Schuyler et al. 2014a, 2014b; Schuyler 2014). This work is expansive and builds on previous reports of debris ingestion to analyze geographic patterns and the similarity of certain debris items to the natural prey of sea turtles. Authors determine a significant increase in the probability that green and leatherback turtles will ingest debris over time, and oceanic species are at the greatest risk of lethal and sub-lethal effects caused by ingested debris (Schuyler et al. 2014a).

Debris Ingestion: Mammals. In two papers published in 2014, Madeira Di Beneditto and Arruda Ramos (2014) and Madeira Di Beneditto and Rodrigues Awabdi (2014) provide information about debris ingestion in large marine vertebrate species in coastal southeastern Brazil. They noted higher concentrations of debris in benthic environments and suggest benthic feeders may be at higher risk for ingesting debris (Madeira Di Beneditto and Awabdi (2014). Similarly, differences in feeding habits were noted to drive variances in debris ingestion between coastal dolphin populations of *Pontoporia blanvillei* and *Sotalia guianensis*, with the ingestion rate of the benthic feeder *P. blanvillei* was estimated at 15.7% of individuals examined, while only 1.3% of *S. guianensis* individuals had debris items within stomach contents. Kaladharan et al. (2014) describe the necropsy of a Longman's beaked whale (*Indopacetus pacificus*) that stranded near Sutrapada, India and upon examination died after the ingestion of four thick plastic shopping bags (190 g) that blocked the intestinal tract. Authors link this to the locally high incidences of anthropogenic litter. Continuing work on marine vertebrates in the Mediterranean, Fossi et al. (2014) measure higher concentration of phthalates in the muscle of basking sharks (*Cetorhinus maximus*) than in blubber of the Mediterranean fin whale (*Balaenoptera physalus*) and discuss possible connections with leaching of these chemicals from fragments of plastic debris.

Alien and Invasive Species Transport*.* Marine debris has been documented as a vector for the transport of various organisms that attach to debris as a hard substrate. There is much unknown about the frequency of organisms rafting on marine debris and the incidences of aquatic invasions by non-indigenous species. Building on the current knowledge base, rafting organisms are documented in Calder et al. (2014). Scientists note the presence of fourteen species of hydroids attached to debris that originated in Japan after the Tohoku tsunami in 2011 and stranded on coastlines in the northwestern United States. Three species of hydroids were noted as cryptogenic – it is unknown if they are native or exotic to the Pacific coast of North America (Calder et al. 2014). The authors conclude by underscoring the potential importance of large debris items, such as the docks that floated from Japan to the Pacific coast of North America, in providing sufficient substrate to host communities of organisms in global dispersal. Also in the North Pacific, Goldstein et al. (2014) demonstrate a positive relationship between the size of a debris "raft" and the diversity of taxa associated and supported. Potential causes for this relationship include stochastic settlement of biological taxa randomly on all debris items present within the system, as well as increased migration onto debris items that outpaces the rate of extinction. Larger debris objects may have an advantage in greater stability at the water's surface, in addition to presenting a greater surface area on which biological settlement may occur (Goldstein et al. 2014).

Oberbeckmann et al. (2014) exposed polyethylene terephthalate bottles to conditions in the North Sea for six weeks, and determined diverse microbial communities within the resulting biofilms that varied by season and specific location. Harrison et al. (2014) investigated the colonization of low-density polyethylene in benthic habitats through the use of mesocosm experiments. Their study showed significant increases in bacterial concentration after seven days and shifts in bacterial community composition with sediment type that did not persist over the experiment, given the dominance of *Arcobacter sp*. and *Colwellia sp*. after fourteen days (Harrison et al. 2014). This study's focus on benthic colonization stands in contrast to most of the literature, which has focused on floating pelagic plastic. Another example of such a study highlighting debris as a new type of pelagic habitat is Reisser et al. (2014), which characterizes biological diversity on millimeter-sized floating plastic debris from coastal and oceanic Australian waters. Importantly, scientists investigated the physical appearance of the debris particles through scanning electron microscopy and observed surface textures including pits and grooves similar to the shape of organisms such as diatoms and coccolithophores that colonized the debris (Reisser et al., 2014). One potential conclusion is that some organisms colonizing the plastic are able to degrade it.

In a study to document and monitor potential habitat changes caused by marine debris, Taylor et al. (2014) evaluated the deep-sea communities that associate with a lost shipping container abandoned to the seafloor, noting the faunal assemblages associated with the container and benthos within 10 meters of the container were significantly different than the surrounding area, possibly resulting from the slow successional rate for crinoids, sponges, and soft corals at that depth Interestingly, though the container is a disturbance to the seabed which alters local flow, it also increases habitat heterogeneity and hard substrate on which megafauna may associate.

Chemical Uptake. Scientists investigating marine debris and its role in the transfer of chemicals to the environment and marine organisms are increasingly focusing on smaller particle sizes that may provide greater surface area and greater likelihood of ingestion to a wide range of organisms. In Rossi and Monticelli (2014), the direct ingestion of plastic nano-particles is considered in the context of understanding the interaction of these particles with lipid membranes. Such studies are important within the medical field, and Rossi and Monticelli (2014) focus research on simulating molecular interactions between medical applications (e.g., dendrimers and linear charged polymers) and potential polymers of debris such as polystyrene. Characterizing such interactions will improve understanding of how chemicals may leach from plastic debris particles.

Bakir et al. published two papers in 2014 to further investigate desorption kinetics of persistent organic pollutants (POPs) under laboratory conditions that simulate the natural physiology of marine organisms, as well as the sorptive behaviors of POPs to synthetic polymers and how these behaviors are affected by varied salinity regimes (Bakir et al. 2014a, 2014b). Authors concluded that salinity (tested at 0, 8.8, 17.5, 26.3, and 35 parts per thousand) did not have an effect on the distribution coefficients governing sorption of phenanthrene (Phe) and DDT to the synthetic polymers polyvinyl chloride (PVC) and polyethylene (PE) (Bakir et al. 2014b). Salinity did not affect time to reach equilibrium, desorption rates, or capacity for accumulation of Phe onto the polymers; however, there was a slight decrease in sorption capacity for DDT onto the polymers that was influenced by increasing salinity. Bakir et al. (2014b) determined that sorptive dynamics are governed more by the combination of contaminant and polymer type than by salinity, and developed a model that estimates between 5 and 15 days for contaminants sorbed to plastics in estuaries to leach from the polymer such that the polymer attains lower contaminant concentrations once it reaches marine waters. A second study noted enhanced desorption of POPs to synthetic polymers under simulated gut conditions than compared to sorption of contaminants in water alone (Bakir et al. 2014a). This study has implications for risk exposure and assessment models that address the impacts of plastic particle ingestion to marine organisms. Simulated gut conditions for warm-blooded organisms provided the most favorable environment for desorption kinetics (Bakir et al. 2014a). Addressing similar knowledge gaps, Lee, Shin and Kwon (2014) used laboratory experiments to measure partition coefficients between microplastic particles and seawater for 8 polycyclic aromatic hydrocarbons (PAHs), 4 hexachlorocyclohexanes (HCHs), and 2 chlorinated benzenes (CBs), using three polymers that represent a large percentage of debris found in the marine environment (PE, PP, PS). Partition coefficients were derived using a third phase (in this case, polydimethylsiloxane) to speed the process, ranged from 2.04-7.87 for polyethylene and a similar range for other polymers, and showed general agreement with log Kow (octanol-water partition coefficient) values.

Polychaete worms (*Arenicola marina)* were exposed to microplastics coated with nonylphenol and phenylrene and the added chemicals Triclosan and PBDE (Browne et al. 2014). Nonylphenol was taken up by the worms, reducing the ability of the coelomocytes to remove pathogenic bacteria. Uptake of Triclosan affected the worm's ability to burrow in the sand and death. The results indicate that micoplastics can affect the physiology of benthic organisms .

Utilizing field collections, Kwon et al. (2014) analyzed the regional distribution of styrene constituents generated from polystyrene (PS; foamed plastic) degradation in the North Pacific, and found breakdown products were more likely to exist along the North American west coast than in Alaska or Hawaii though drivers of this trend are unknown. Llorca et al. (2014) detected perfluoroalkyl substances associated with a limited number of plastic pellets and sediment samples collected along coastlines in Greece. While these studies do not explicitly address the effects of the plastics and/or the associated contaminants, they assist the scientific community in more fully understanding the inherent risks posed by plastic particles as transport mechanisms for chemical contaminants, including metals, plasticizers and other chemicals integrated into the polymer, and contaminants sorbed from surrounding waters or sediments.

Sorption and concentration of metal contaminants onto synthetic polymers has not been well documented. A paper by Holmes et al. (2014) is one of several studies published in 2014 that address this knowledge gap. One primary consideration when assessing trace metals associated with a plastic matrix is the methodological techniques used within laboratories that consistently rely on the use of plastic equipment and materials when analyzing metals in environmental samples. Method development would seem to be in early but promising stages. Holmes et al. (2014) determined greater sorption to beached polyethylene pellets than virgin pellets for all metals assessed (cadmium, cobalt, chromium, copper, nickel, and lead), and reported various shifts in adsorption based on salinity regime and pH though adsorption is less likely to occur onto synthetic polymers than to sediments. Rochman, Hentschel and The (2014) assessed dynamics of metal accumulation onto five synthetic polymers (polyethylene terephthalate (PET), high-density polyethylene (HDPE), polyvinyl chloride, low-density polyethylene (LDPE), and polypropylene (PP) during a year-long exposure experiment in San Diego Bay. Trace metal sorption did not significantly differ among the five polymers, though all metal concentrations increased during the twelve-month study which implies persistent debris fragments accumulate trace metals over time. In a new study by Graca et al. (2014), researchers analyzed virgin and beached foamed plastics for total inorganic mercury content and found an order of magnitude higher mercury content in foam debris (5.20 ng/g) over virgin foam pieces (0.23 n/g). The presence of biofilms and the interaction of the particles with solar radiation lead to seasonal peaks in Hg concentration during summer months (Graca et al. 2014). Authors noted occasional foam debris particles exceeded bottom sediment and soil standards by measuring approximately 3,000 ng/g mercury, and suggest that with age these particles could be enhanced in their ability to transport mercury in the environment (Graca et al. 2014).

Researchers continue to use a combination of modeling, laboratory, and field conditions to develop a better understanding of the factors affecting sorption kinetics, and the connection between sorption and desorption upon particle ingestion. A modeling approach assessed the leaching of plastic additives to marine organisms by applying a biodynamic model to calculate potential for transfer of nonylphenol (NP) and bisphenol A (BPA) to intestinal tissues of the lugworm (*Arenicola marina*) and North Sea cod (*Gadus morhua*) (Koelmans et al. 2014a). Public concern over the leaching of such contaminants to seafood emphasizes the importance of the study's major finding that plastic ingestion is a negligible source of plastic additives to cod when compared with other pathways, though plastic particle ingestion could be a substantial pathway for introducing these chemicals to the lugworm.

Nilsen et al. (2014) propose to characterize ingested plastic particles by so-called "indicator chemicals" sorbed to the particle, in order to better document particles and fragments that are not able to be characterized as a particular polymer or resin type. According to this analysis, approximately 68.4% of unidentifiable plastic fragments ingested by Laysan albatross from Kure Atoll are polypropylene, while 20.5% are polyvinyl chloride (Nilsen et al. 2014). Authors suggest that infrequently recycled plastic resins are the more prevalent in this study and potentially more persistent in the environment (Nilsen et al. 2014). Resin type may influence and govern risks posed by debris ingestion, especially as risk relates to the leaching of chemicals within the digestive tract, and thus this approach to identifying indicators of resin type may prove useful in future risk calculations.

Rochman, Kurobe, Flores and Teh (2014) built on previous work to investigate the potential for endocrine disruption in Japanese medaka (*Oryzias latipes*), a common organism used in toxicity testing, derived from exposure to plastic marine debris. Dietary exposures reveal altered gene expression in male and female fish exposed to polyethylene pellets that had spent three months in San Diego Bay, including down-regulation of choriogenin in males and vitellogenin in females, which the authors suggest warrants further investigation. Another study tested the hypothesis that chemical concentrations of plastic particles would be greater in areas predicted to be debris "hotspots" by a previously published predictive oceanographic model (Rochman et al. 2014). The study involved field collections to examine accumulation of POPs in myctophid fish inhabiting pelagic waters in the South Atlantic Ocean, as well as to document debris concentration in this region. Contaminant concentrations in the fish varied without respect to debris density except for several higher-brominated congeners of polybrominated diphenyl ethers (PBDE) measured in fish tissue (Rochman et al. 2014c). Results highlight the difficulty in determining the relative contribution of various sources of persistent organic pollutants and trace metals to marine organisms; it is challenging to determine relative contributions of chemicals sourced from marine debris versus other exposure pathways, as tissue residues reflect integration of inputs over long time frames and large spatial scales.

Oil Spills: The Deepwater Horizon

Major oil spills, the Deepwater Horizon (DWH), Helbei Spirit, Prestige, and Exxon Valdez, continued to generate a research long after the events. This section explores papers stemming from the 2010 DWH incident.

Reviews. Pennings et al. (2014) conducted a literature review of effects of oil exposure to terrestrial arthropods and provided research topics for future studies. Field observations conducted at sites impacted by Deepwater Horizon showed an initial decline in terrestrial arthropods and a fast recovery of healthy populations. The authors suggest further investigations into the relationship of arthropods and their associative habitats with various floras in conjunction with their resilience to quantity of oiling.

Özhan et al. (2014) conducted a literature review on the effects of crude oil to phytoplankton species. The paper cites phytoplankton EC50's range from 1.03->50mg/L when exposed to crude oil and 1.01-1031mg/L for crude oil compounds. The authors also provide citations regarding phytoplankton community diversity shift toward higher concentration of petroleum degrading microbes as a response to exposure to oil from the Deep Water Horizon event.

Exposure Concentrations. Two years after the Deepwater Horizon oil spill, Bianchi et al. (2014) identified dissolved organic carbon (DOC) concentrations (284 ± 64 μM; $n = 3$) and oil-derived chromophoric dissolved organic matter (CDOM) signatures at certain stations in the deep waters of the Gulf of Mexico that remain higher than typical deep water values ((ca.40−50 μM).

Gray et al. (2014) used liquid chromatography/tandem mass spectrometry with isotope-dilution quantification to measure concentrations of dioctyl sulfosuccinate (DOSS) at the surface and in the water column in the vicinity of the Macondo (a.k.a. Deepwater Horizon) well head from May 27-June 4, 2010 during the Deepwater Horizon oil spill. The highest concentrations of DOSS were recorded within 3 km of the wellhead, but only three sites had concentrations which exceeded the U.S. Environmental Protection Agency's aquatic life benchmark of 40 μg L^{-1}.

Smith, Flemings and Fulton (2014) studied the transport of hydrocarbons from two deep water vents in the Gulf of Mexico incorporating salinity and temperature gradients to include multi-phase hydrocarbon distribution. The author's one dimensional advection-diffusion model predicted hydrocarbon flux to exceed over 100 times previous estimates and attributes natural seeping hydrocarbons supporting bacteria communities that contributed to the high biodegradation rates observed during the Deepwater Horizon event.

Microbial Community Response. MacDonald et al. (2014) introduced the special issue of Environmental Research Letters which contains papers related to the Deepwater Horizon oil spill and covers topics ranging from the transience and persistence of discharged hydrocarbons to microbial responses to the large input of oil derived carbon.

Crespo-Medina et al. (2014) tracked methanotrophic activity in the deep sea during and after the Deepwater Horizon oil. They found that the methanotrophic population peaked in May/early June 2010, but then crashed in late June, even though deep water methane concentrations remained elevated (tens of μM,with a maximum of 180μM) above background. Since methane remained abundant, the authors suggest that physiological or environmental factors such as kinetic selection of the methanotroph population, trophic interactions or mortality (for example, viral lysis, selective grazing pressure), nutrient or trace metal limitation are among the likely causes.

Kostka et al. (2014) discussed the impact that cutting edge molecular and biogeochemical techniques (including high throughput sequencing, isotope tracers, and _omic approaches) are having in advancing understanding of the biogeochemical processes and metabolic pathways that control hydrocarbon biodegradation in marine systems. They then outlined the major themes of the papers contained within the Research Topic, including diversity of and impacts to the microbial community in the Gulf of Mexico from the Deepwater Horizon oil spill, bridging laboratory studies of biodegradation to the field, and environmental controls of oil biodegradation in marine sediments.

Engel and Gupta (2014) found that the microbial community of sandy beaches in Grand Isle, Louisiana and Dauphin Island, Alabama that were impacted by oil from the Deepwater Horizon oil spill shifted from bacterial communities that are associated with fecal contamination (pre-oiling) to open ocean associated communities known to consume hydrocarbons (post-oiling). The authors suggest that sand washing and tilling caused the regime shift.

Lamendella et al. (2014) reported a significant increase in microbial cell density and a shift in the microbial community toward a hydrocarbon degrading consortium on a beach in Louisiana after it was heavily contaminated with oil from the Deepwater Horizon oil spill. 16SrNA gene sequencing and Metatranscriptome profiling established the presence of both known and suspected hydrocarbon degrading microorganisms including Marinobacter, Roseobacter, and Pseudomona species, among others.

Scott et al. (2014) compared sediments impacted by the Deepwater Horizon event, sediments exposed to natural seeps found in Santa Barbara, and reference sediments collected in Gulf of Mexico to assess the impact of polyaromatic hydrocarbon exposure to the nitrification processes in the upper layer of marine surface sediments. The authors specifically measured effects on genes involved with nitrogen metabolism and nitrogen cycling and reported significant differences between the three sediments in relative turnover rates for nitrogen metabolites.

Thomas et al. (2014) conducted a 3 month microcosm study using field collected Apalachicola Bay, Florida oysters, water and sediments, as well as, including oil-degrading bacteria colonies inoculated with oil from the Deepwater Horizon spill site. The authors report changes in the diversity of bacteria in the microcosms with respect to the biodegradation process of oil, however, they report there was not a reference "no oil" microcosm to compare the results.

Joye et al. (2014) reviewed recent literature outlining the effects of the Deepwater Horizon oil spill on microbial community composition and activity in various Gulf of Mexico ecosystems. The microbial community shifted rapidly in response to the oil and gas input, a wide variety of microorganisms, including alkane, PAH, and methane degraders and nitrifying microorganisms responded, marine oil snow formed, and dispersants appear to have had an adverse effect on the microbial community and microbe-based food web, among other effects.

Effects on Flora. Judy et al. (2014) reported that common reed (*Phragmites australis*) was resilient to oiling of its shoots (0-100% cover) by both weathered and emulsified Macondo oil from the Deepwater Horizon spill, but that oil applied to the soil (0-16 l m-2) and repeated oiling of shoots resulted in reductions in above-and belowground plant growth.

Effects on Phytoplankton. Özhan, Parsons and Bargu (2014) reviewed the factors that influence the toxicity of oil to phytoplankton which includes among many other things, diversity of phytoplankton physiologies, geographic location, oceanographic and meteorological conditions, nutrient concentrations, seasonal variations, oil type, and oil dosage (impacts generally seen between 1 and 100 mg/l), and outlined direct and indirect (e.g. formation of oil films that limit gas exchange and reduce light penetration) toxic effects. The authors then summarized recent studies of effects in the phytoplankton community from the Deepwater Horizon oil spill including phytoplankton blooms and the formation of marine snow, as well as laboratory studies that examined the toxicity of Macondo oil and dispersants.

Effects on Mollusks. Using stable carbon (C14) analysis of the tissues of barnacles (*Balanus sp*.) and marsh mussels (*Geukensia demissa*), Fry and Anderson (2014) concluded that little to no oil from the Deepwater Horizon oil spill was incorporated into estuarine food webs. The authors also measured respiration rates in estuarine waters that were within the range of previously reported values for uncontaminated waters, leading them to conclude that there was very little microbial community enhancement in estuarine waters as a result of the oil spill.

Synder et al. (2014) monitored PAH concentrations in field collected Coquina clams (*Donax spp*) and shoreline sediments for 21 months. The authors report PAH concentrations in clams ranging from 5-1000ppb and sand 5-500ppb. They suggest clams are a useful tool to monitor PAH pollution in high energy surf zones.

Effects on Crustaceans. Yednock and Neigel (2014) field collect blue crabs, *Callinectes sapidus* from Louisiana and Texas coastline during May to July 2010 and 2011 to conduct genetic sequencing analysis across 9 sample locations. Significant differences were reported for adult individuals and the authors speculated results were due to exposure to oil and Corexit 9500A mixtures during the DWH event coupled with the shrimp fishery closures that would predictably negatively impact adult blue distribution.

Effects on Pelagic Fish. Brette et al. (2014) demonstrated that crude oil from the Deepwater Horizon oil spill affects the regulation of cellular excitability in cardiomyocytes isolated from juvenile bluefin (*Thunnus orientalis*) and yellowfin (*T. albacares*) tuna. While effects were seen at ΣPAH concentrations as low as 4 µg/liter, the cardiotoxicity of the oil correlated to the concentrations of three-ringed PAHs in the oil samples.

Incardona et al. (2014) found adverse effects on cardiac function in Bluefin tuna, Yellowfin tuna, and Amberjack (*Seriola dumerili)* fish embryos at PAH exposure levels (1-15µg/L total PAH) consistent with those documented in the Gulf of Mexico during the Deepwater Horizon oil spill.

Mager et al. (2014) reported that juvenile mahi-mahi *(Coryphaena hippurus)* displayed impaired swimming performance when exposed to the water accommodated fraction of oil from the Deepwater Horizon oil spill. Embryos/larvae that were exposed for 48-hour at 1.2±0.6 µg L-1 ΣPAHs and then raised to juveniles displayed reduced swimming efficiency as a latent effect, while exposed juveniles displayed such effects only at the highest concentration tested (30±7 µg L-1ΣPAHs).

Effects on Nearshore Fish. Crowe et al. (2014) exposed Gulf killifish (*Fundulus grandis*) to water-accommodated fractions (WAFs) of crude oil (7.0 _0.10 mg/L C6-C28 at time 0) from the Deepwater Horizon oil spill for 12, 24, and 48-hour intervals. They observed a significant increase (66%) in the expression of the CYP1A in the fish exposed for 24-hours, and significant increases in antioxidant capacity of nonenzymatic antioxidants in exposed fish at each time point.

Murawski et al. (2014) investigated reports of offshore fishes exhibiting skin lesions following the Deepwater Horizon well blowout and documented a significant decline in overall lesion frequency (1.9% to 0.9% (all species sampled) and bile PAH levels (Red Snapper (*Lutjanus campechanus*) between 2011 and 2012 sampling events at the Northern Gulf of Mexico sampling sites. There was a strong correlation ($r2=0.82$; $p<0.001$) between oil collected at the Deepwater Horizon wellhead and the composition of PAH parent compounds and alkylated homologs in the Red Snapper liver samples.

Seafood Safety. Fitzgerald and Gohlke (2014) sampled seven species of Gulf of Mexico reef fishes for PAHs, Dioctyl Sodium sulfosuccinate (a component of the dispersants Corexit 9500A and 9527A), and several metals in the aftermath of the Deepwater Horizon spill as part of a fishing industry led study to ensure the safety of their catch. Of 92 samples, none exceeded federal safety standards for benzo(a)-pyrene-equivalents or contained DOSS, and metals were absent or within expected values.

Effects on Deep Sea Coral. Fisher et al. (2014a) documented adverse impacts attributable to the Deepwater Horizon oil spill to two additional deep sea coral communities located six kilometers and 22 kilometers away from the well-head. They also surveyed numerous coral communities around the northern Gulf of Mexico, and found no acute impacts to corals at depths between 400 and 850 km and greater than 30 km from the well-head. Fisher et al. (2014b) reviewed the effects of oil from the Deepwater Horizon oil spill on corals and soft-sediment ecosystems in the Northern Gulf of Mexico below 400 m. Damage to coral colonies in three separate coral communities was easily observed by the presence of dead/dying branches that had been colonized by hydroids. Examination of sediment cores from areas with 30km of the Macondo (Deepwater Horizon) well site contained significantly higher levels of saturated hydrocarbons and PAHs (459 – 47,600 µg/kg) and less macrofaunal and meiofaunal diversity than previously recorded in the Gulf of Mexico (0-1030 µg/kg), although distribution was patchy. Coral associated sediment communities displayed similar decreases in diversity and shifts in community structure.

Fredericq et al. (2014) documented a dramatic die off of rich algal assemblages (rhodoliths) after the Deepwater Horizon oil spill at two previously documented sites in the northern Gulf of Mexico. They speculate that the corals may have suffered “chemical bleaching” from the possible induction of oil into porous bedrock, stimulating anaerobic sulfate reducers and an associated production of hydrogen sulfide, or from a potential increase in their release of dimethylsulfoniopropionate production. Bare rubble brought back from the sites was naturally recolonized in the laboratory setting, while the same rubble at the sampling sites had not recovered as of October 2013.

Effects on Birds. Bergeon Burns et al. (2014) reviewed the effects of oiling on terrestrial vertebrates which may include reduced water repelling properties, thermoregulation difficulties, immunosuppression, habitat alteration and prey impacts with subsequent effects on the organisms, reduced reproductive success, and compromised ability to handle additional stressors, among others. The authors then outlined their approach in attempting to determine potential effects of oil from the Deepwater Horizon oil spill on seaside sparrows (*Ammodramus maritimus*) and marsh rice rats (*Oryzomys palustris*) living in heavily oiled salt marsh in Louisiana, while noting the difficulty of distinguishing the effects of the oil from many other complex and potentially confounding factors present in the Gulf of Mexico (e.g. hurricanes, subsidence, erosion, development pressure, salinization, other variability introduced by the presence of oil (e.g. pathogen increases) etc.). The authors' referenced preliminary data that shows seaside sparrow nests are significantly less likely to fledge in oiled areas than are those in non-oiled areas, but have not yet analyzed the data in the context of other confounding factors.

Franci et al. (2014) compared body mass, corticosterone, and prolactin levels in two populations of Northern gannets (*Morus bassanus*): one which overwinters in the Gulf of Mexico and potentially was exposed to oil from the Deepwater Horizon oil spill and one that overwinters on the Atlantic coast. The authors found no difference in the populations, and no variation in the hormone levels over the early to late incubation period.

Paruk et al. (2014) field collected 38 common loons (*Gavia immer*) off the coast of Louisiana near Barataria Bay during Jan-March of 2011 and 2012 and extracted blood samples along with biological measurements prior to release. In 2011, 3 of the 17 birds captured tested positive for Anthracene with concentrations ranging from 1.7-2.3 ng/g. In 2012, 13 of the 21 birds captured tested positive for Anthracene concentrations ranging from 3.4-8.0ng/g and 1 tested positive for Fluoranthene at a concentration of 1.6ng/g.

Using an exposure probability model incorporating oil slick size, bird density, and proportionate mortality for their calculations, Haney et al. (2014) estimated that between 36,000 and 670,000 birds died in the offshore (+40km) of the Gulf of Mexico (GOM) as a result of exposure to oil from the Deepwater Horizon oil spill.

Using both a carcass sampling model and an exposure probability model, Haney et al. (2014b) calculated that approximately 700,000 birds died in the coastal GOM as a result of the Deepwater Horizon oil spill, but the numbers may be as low as 320,000 and as high as 1,900,000 within the 95% uncertainty bounds.

Walter et al. (2014) banded over 1,000 Brown Pelicans (*Pelecanus occidentalis*) between 2007 and 2009 and observed their migration and age distribution at study sites between 2008 and 2011. The authors observed 54 pelicans in 2011 after the DWH event and specifically 12 of the 182 banded pelicans that underwent oiling, rehabilitation, and release. The authors could not attribute that age structure variation was a cause of exposure to oil from the DWH event.

Effects on Marine Mammals. Schwacke et al. (2014) conducted an observational study on bottlenose dolphins (*Tursiops truncatus)* using Barataria Bay, LA, a heavily oiled location impacted by the Deepwater Horizon event, and Sarasota, FL as a reference site. Dolphins were captured, evaluated, and released. The authors reported dolphins in Barataria Bay showed common symptoms due to oil exposure, such as lung disease, hypoadrenocorticism, and pulmonary consolidation as well as mortality rates higher than previously recorded. A critique and reply followed publication.

Rice et al. (2014) collected acoustic recordings of the Bryde's whale (*Balaenoptera edeni*) in the Gulf of Mexico between 2010-2012. The authors suggest the existence of the small group of whales in the Gulf of Mexico and should be considered before dispersant use.

Using a Bayesian modeling approach that incorporated a conditional occupancy estimator and prior information on detection probability, Martin et al. (2014) estimated that at the time of their aerial surveys, fewer than 2.4% of the manatee (*Trichechus manatus latirostris*) habitat potentially impacted by oil from the Deepwater Horizon oil spill may have contained manatees, and probably contained 107 or fewer manatees.

Use of –Omics Methods.. Bik (2014) reviewed how the use of metagenomics is revolutionizing our understanding of ecosystem functioning. _Omics studies in the aftermath of the Deepwater Horizon oil spill were cited to illustrate how the use of –omics data is increasing our understanding of biological responses to the influx of oil in the Gulf of Mexico.

Other Oil and Fuel Spills and Research

Spain (Prestige Spill). Barros et al. (2014) documented long term reproductive impairment (+10 years) in the European Shag (*Phalacrocorax aristotelis*) as a result of oil exposure from the 2002 Prestige oil spill. Reproductive success was reduced by 45% in oiled colonies vs. unoiled colonies after the spill as compared to the years prior to the spill. Castège et al. (2014) examined the effects of the Prestige oil spill on the benthic community of the rocky shore of Guéthary (south of the Bay of Biscay, France). Three years post spill, taxonomic richness had returned to pre-spill levels, but the community structure took almost five years to reestablish. Junoy et al. (2014) found that although macrofaunal assemblages of O Rostro Beach (Galicia, NW Spain), the beach most heavily oiled by the Prestige oil spill, displayed negative effects (reduced abundance and species richness) in the first six months after the spill, the community had recovered to pre-spill conditions by 2004.

South Korea (Hebei Spirit oil spill). Two years after the cleanup of the 2007 Hebei Spirit oil spill, Kim, Hong, Kim and Yang (2014) found that PAH levels in bottom sediments both inside and outside of Mohang Harbor, Korea ranged from 24-366 µg/kg, posing minimal ecological risk. Lee, Kim, Jeong et al. (2014) documented chamber breakage in 71.6% of the benthic foraminifera, *Ammonia beccarii*, collected from the substrata sediment of Sogeunri tidal flat, Taean Peninsula, Korea. The authors attributed the breakage to decalcification caused by low pH (6.98) of the sediments contaminated with oil-mineral aggregates deposited from the Hebei Spirit oil spill. After three years post-spill monitoring Seo et al (2014) reported that subtidal sediment PAH concentratios returned to below background. There were moderate changes in the benthic infauna community, and no mass mortlities of amphipods except at two stations. The opportunistic colonial polychaete, *Prionospio paradisea* dominated one site 10 moths after the spill.

India (Ship collision). Sukumaran et al. (2014) conducted a 15 month environmental monitoring study to assess the impact of 800 tons of fuel oil spilled during a ship collision between MSC Chitra and MV Khalija 3 in Mumbai Harbour in 2010. The authors collected sediment cores for macrobenthic organism analysis, sediment characteristic and hydrocarbon concentration. Macrobenthic organsims showed less diversity after the oil spill than the next year and the authors reported a range of hydrocarbon concentration in sediments to be 5.6-311.3 μg/g.

Alaska (Exxon Valdez). Fukuyama et al. (2014) compared intertidal infaunal communities in unoiled, oiled and untreated or lightly treated (oiled), and oiled sites treated with high-pressure hot water washing (treated) over the period of 1990-2000 to evaluate impacts and recovery after the Exxon 1989 Valdez oil spill. They found that the infaunal communities in the oiled and treated sites had largely recovered by 2000, but treated sites were lacking large sized little neck clams (*Leukoma* (*Protothaca*) *staminea*) found in untreated and oiled sites, correlating to fewer fine grained sediments in the treated sites related to cleanup activities. Harwell and Gentile (2014) used new estimates of subsurface oil residue encounter rates for sea otters (*Enhydra lutris*) at northern Knight Island to update their individual-based model risk assessment. The authors concluded that the risk to sea otters from these encounters is small, and that the population has fully recovered from the Exxon Valdez oil spill.

San Francisco (Dubai Star). Hwang, Stanton, McBride and Anderson (2014) found that body burdens of PAHs and levels of lysosomal membrane destabilization in mussels were higher from animals collected from moderately oiled shorelines than those collected from the same shoreline prior to oiling from the 2009 Dubai Star release (average 1077±519 ng/g (dry weight) vs up to 87,554 ng/g post spill). Body burdens of PAHs had returned to pre-oiling levels three months post-spill.

Arctic. Dunton et al. (2014) sampled the benthos of the northeastern Chuckhi Sea to establish baseline conditions for chemical and biological characteristics. Metals, PAHs, and aliphatic hydrocarbons were at natural background levels at nearly all sites sampled, and there is a rich benthic community of organisms with a complex food web.

Jörundsdóttir et al. (2014) documented concentrations of non-alkylated PAHs and inorganic trace elements (arsenic, cadmium, mercury, lead) in blue mussels (*Mytilus edulis*) and PAH metabolites in cod (*Gadus morhua*) bile in the Nordic artic and sub-Arctic coastlines to establish background conditions. In general, PAH concentrations and inorganic trace elements were low in mussels (Σ16PAH ranged between 28 and 480 ng/g d.w.) and 1-OH-pyrene was only quantifiable (between.44 and 140 ng/ml bile) in samples collected from the Norwegian coast. Harvey et al. (2014) documented very low concentrations of PAHs (<1600 ng g-1 dry wt) in the top 0-1 cm of sediments of the Chukchi Sea, and found that common Arctic cod (*Boreogadus saida*) CYP1A1, GST, and SOD enzyme levels were comparable to baseline levels in other pristine systems. They also opportunistically sampled Northern whelk (*Neptunea heros*), finding lower concentrations of PAHs (4.5–10.7 ng g-1 wet wt), but larger concentrations of aliphatic n-alkanes (C19-C33) (0.655–5.20 µg g-1 wet wt) in larger animals as compared to smaller animals.

Payne et al. (2014) conducted 14-day multi-species community toxicity tests and 7-day single species toxicity tests using field collected Antarctic zooplankton *Oncaea curvata, Oithona similis*, and *Stephos longipes* exposed to Special Antarctic Blend diesel water accommodated fractions. Three replicates of zooplankton community exposed to SAB diesel WAF treatments reported LC50 values ranging from 186-1091 µg TPH/L. The authors reported LC50 values of 158, 176, 188 µg TPH/L *for O. curvata*, *O. similis* and *S. longipes*.

General. Reviews. Chang et al. (2014) reviewed the consequences of oil spills from tankers and developed an overview framework to be used as a basis for planning and discussion by focusing on the following areas: the oil spill itself, disaster management, the physical marine environment, marine biology, human health, economy, and policy. The authors then used the expected increase in tanker traffic in Vancouver, Canada as a case study to illustrate potential impacts of a spill and their associated complexities, including the need to account unique conditions associated with each spill and locality.

Farrington (2014) outlined the fate and effects of oil spills in the marine environment and discussed some cleanup and mitigation measures, including the use of dispersants, Environmental Sensitivity Indices, and oil spill models.

Kirby et al. (2014) outlined the following eight principles of effective post-oil spill monitoring: scientific guidance, skills and knowledge, equipment, funding, responsibility and management, integration and coordination, support and buy-in, and practice. The authors then provided a case study of the United Kingdom cross-government program known as Premiam (Pollution Response in Emergencies: Marine Impact Assessment and Monitoring) which illustrates the implementation of the principles.

Klemas and Blažauskas provided a brief overview of information and actions needed to minimize the damage from oil spills and facilitate cleanup efforts, including the use of environmental sensitivity maps. The authors then introduced the Baltica Special Issue publication that stemmed from the common Lithuanian-Russian project, "Development of solutions for effective oil spill management in the South-Eastern Baltic."

Ventikos and Sotiropoulos (2014) provide a literature review on the costs of responding and cleaning oil spill incidents and statistical models from a data set including 107 spills ultimately illustrating size and location directly influence costs. The authors also categorize accident types: sinking, grounding, collision, other and quantity spilled 1-10,000tons to provide cost estimates of impact. The authors do not take in to account type of oil spilled and location.

Suzdalev et al. (2014) summarized the increase of shipping and potential threat of oil spills in the Baltic Sea. The authors provide examples of past spills in the area of concern and highlight improvements of spill response planning that have occurred in the area including area contingency plan updates, updating sensitive shoreline mapping tools, and use of new oil fate modelling tools.

Peters and Siuda (2014) reviewed the history of tar balls and tar mats observed in the Sargasso Sea from 1960-present and asserted detectable levels of tar balls have declined over time due to increased regulations and environmental monitoring.

Laboratory and field investigations. Han et al. (2014) found that exposure to the water accommodated fraction of Iranian crude oil at 20%, 40%, 60%, and 80% of stock solution 25 g crude oil/L caused significant delays in development, molting, and hatching rates in the copepod *Tigriopus japonicus*, as well as increased the activities of antioxidant enzymes. There was no mortality, even at 100% WAF exposure. The authors also identified three TJ-CYP genes as potential biomarkers of oil exposure.

Harms et al. (2015) found significant increases in key hematological values in loggerhead turtle (*Caretta caretta*) hatchlings exposed to Gulf Coast mixed sweet crude oil (0.833 mL/L) and/or dispersant (Corexit 9500A, 0.083 mL/L) as compared to non-exposed controls, with the greatest changes in the animals exposed to combined oil/dispersant mixtures.

Jurelevicius et al. (2014) dosed marine (Massambaba Beach) and hypersaline waters (Vermelha Lagoon) from the Massambaba Environmental Protection Area, Rio deJaneiro, Brazil with crude oil (1 % v/v), naphthalene as a representative PAH (1 % w/v), or heptadecane as a representative aliphatic hydrocarbon (1 % v/v) to observe the effects on the Archaeal communities. No Archaeal communities were detected in the marine waters contaminated with hydrocarbons, but they were present in the hypersaline hydrocarbon contaminated water.

Lindgren et al. (2014) found that the addition of Swedish Mk-1 diesel at nominal PAH concentrations of 1300 μg ΣPAH/kg dw sediment in three different sediment types (muddy, sandy, and organic) resulted in significant negative impacts to the microbial community as measured by potential nitrification and denitrification, although the negative effect was less in muddy sediments. PAH availability was greater in the sandy and organic sediments than in the muddy sediments. No effect was observed on the meiofaunal community in any of the sediments.

Kang et al. (2014) reported significant declines in the density of meiofauna in all plots treated with 0.125 L, 0.25 L, 0.625 L, or 1.25L of crude oil, and increases in nematode species diversity in all treated plots except the 1.25L treatment as compared to the control plots with no treatment. However, the community recovered to pre-spill character and structure within one month of the spill.

Leite et al. (2014) found no significant differences in nematode total density, diversity, and community structure as compared between marine diesel oil treated (2500 ml application) and control plots located in unvegetated tidal flats of the Paranaguá Estuarine Complex (Southern Brazil). Total aliphatics in experimental plots ranged from 2.31 to 30.8 μg g-1 dry sediment in the treated plots as compared to 1.42 to 2.77 μg g-1 in the control plots.

Morandin and O'Hara (2014) found that feathers exposed to sardine oil sheens (ranging from 0.04 to 3 μm in thickness) displayed significant feather microstructure disruption and significant weight gain from oil and water uptake at all sheen thicknesses tested. From this data and in conjunction with interviews with wildlife rehabilitation experienced with oiled bird rehabilitation, the authors concluded that edible oils are at least as harmful to seabirds as petroleum oil.

Based on data collected from both live and dead oiled birds, Henkel et al. (2014) estimate that >1,000 birds are oiled oil on an annual basis on the California coast by natural petroleum seeps and/or other chronic sources of oil (i.e. shipwrecks Luckenbach and Palo Alto). Oiling peaks in the late winter (Jan.-Apr.).

Stauffert et al. (2014) conducted a 270 day microcosm study evaluating the effect of oil exposure to field collected sediments containing Archaea. The authors quantified Archaea population diversity by using terminal restriction fragment length polymorphism (T-RFLP) fingerprints and conducted 16S rRNA gene and 16S cDNA sequencing. The authors reported oiled treatments had a significant effect on Archaea community structure.

Sugahara et al. (2014) exposed fertilized embryos and larval Takifugu niphobles to pyrene and phenanthrene to assess morphological and behavioral abnormalities due to exposure. Concentrations of pyrene ranged from 10-100ppb and phenanthrene test concentration was 200ppb. The authors reported effect on larval swimming behavior at 10ppb of pyrene and assert the effect was due to morphological stunting in the brain. No effects were observed for phenanthrene exposures.

Witt et al. (2014) conducted Microtox toxicity tests using *Vibrio fisheri* exposed to 128 field collected sediment samples to generate EC50 values and establish data that assists in developing a matrix to identify effects of pollutants in sediments. The authors used EC50 percentages (≤ 1% Very Toxic; 1%<ECO50<2%-Toxic; EC50≥2% - Non-Toxic) to predict and model toxicity of sediments of the Gdansk basin area and reported the toxicity decreased in a trend from the northeast to southwest with median EC50% ranging from 0.63-9.33% for the 5 areas.

Modeling Effect of Oil Exposure. Redman et al. (2014) investigated the bioavailability of hydrocarbons to rainbow trout (*Oncorhynchus myskiss*) using two methods, a PETROTOX model and solid-phase microextraction (SPME) approach. The goal of the research was to assess the appropriateness of using SPME methods to model toxicity and associated pathways as well as identify toxic compounds included in complex heavy fuel oil. The authors suggests the SPME method is a valid simple model to predict toxicity as the values compared appropriately with the values generated by the PETROTOX model.

Vaiene et al. (2014) compared the results of using two bioaccumulation models to predict PAH concentrations in 5 marine organisms: fish species *Gadus morhua*, *Paralichthys olivaceus*, and *Scophthalmus maximus*, the mollusk *Mytilus edulis*, and a crustacean *Pandalus borealis*. The authors investigated the difference between bioaccumulation models that did and did not incorporate ingestion of oil droplets and reported that no difference between models was observed for all species except the mollusk, *Mytilus edulis*.

Risk Assessment. Weller et al. (2014) modeled the effects of multiple stressors on the endangered African penguin *Spheniscus demersus* including anthropogenic influences on food availability, oil spills, and predator/prey dynamics. Data was derived by multiple published studies and expert opinion. The authors reported the results of the model for managing the African penguin indicated preventing oil exposure from incidents coupled with increasing food resources would positively impact population sustainability.

Rogowska et al. (2014) conducted toxicity assays using field collected sediments coupled with a self organizing map algorithim to complete a risk assessment of potential pollution from a shipwreck in Gdańsk Bay. Toxicity tests included *Vibrio fischeri* microtox test, *Sinapis alba* phytotoxicity test, and *Heterocypris incongruens* chronic toxicity test. Sediments were analyzed for PAH and metal concentrations. All available data were utilized in the SOM algorithm which generated a series of similarities between endpoints and pollution concentrations.

Dispersants and Other Oil Spill Treating Agents

Use of chemical dispersants to combat the 2010 Deepwater Horizon (DWH, Gulf of Mexico, USA) oil spill stimulated increased research of the effects of dispersants and dispersed oil on marine and aquatic organisms. Studies in 2014 included effects of dispersant prodcuts and dispersed oils on viruses, bacteria, phytoplankton, zooplankton, and adult or early life stages of mollusks, crustacean, and fishes., and also studies on other oil spill chemical treating agents. Readers should be aware that (1) inconsistent reporting of toxicity in terns of nominal concentrations, measured concentrations and/or mass vs % volume dilutions creates confusion and (2) there remains debate over whether dispersants and oil are synergistic or not.

Viruses. Pham et al. (2014) exposed the aquatic viruses: viral hermorrhagic septicemia virus (VHSV), infectious pancreatic necrosis virus (IPNV), chum salmon reovirus (CSV), and frog virus 3 (FV3) to Corexit 9500 using cell lines propogated from fathead minnow and CHSE-214. The authors reported exposures resulted in no effect for IPNV, reduced infectability at concentrations of .0001-10% v/v in VHSV and FV3, and increased infectability cells at concentrations of 1%v/v and higher in CSV.

Phytoplankton. Diatoms *Isochrysis galbana* and *Chaetoceros sp.*to weathered and non-weathered MC-252 crude oil, and Corexit 9500A to conduct acute and growth inhibition toxicity studies (Garr et al. 2014). No statistically significant effects observed for oil only toxicity tests; however, 96hr IC50 values reported for Chemically Enhance Water Accomodated Fractions (CEWAFs) of non-weathered MC252 crude oil were observed at 149 mg/L and 200 mg/L for *I. galbana* and *Chaetocerous sp* and CEWAFS of weathered oil 48hr test observed at 174 mg/L and 91 mg/L for the same respective species. Özhan and

Bargu (2014) conducted a 10 day microcosm study using field collected phytoplankton as the test organisms and Louisiana crude oil (SLC), Corexit 9500A, and Corexit 9500A treated SLC treatments. The authors reported Corexit 9500A only treatment showed significant effect of growth inhibition at a concentration of 63ppm. The authors had additional test treatments where additive nutrients were applied and observed less of a toxic effect from LSC exposure to specifically sensitive species. Phytoxicity assays were also conducted to assess the effect of exposure to South Louisiana sweet crude oil (SLC) and Corexit 9500A-dispersed SLC on growth of marine diatoms and dinoflagellates (Özhan et al 2014). The authors reported TPH EC50 values for the oil only tests as 2498, 1835, 1751, 1138, 1025 ppb for *Ditylum brightwellii*, *Chaetoceros socialis*, *Pyrocystis lunula*, *Scrippsiella trochoidea*, *Heterocapsa triquetra*, respectively. The Corexit 9500A dispersed SLC treatments EC50 is reported as <100pm for both diatoms and dinoflagellates species.

Planktonic Animals and Life Stages. The active ingredients in Corexit 9500, propylene glycol and 2-butoxyethanol, were evaluated for toxic effects using 48hr acute toxicity methods and the marine foraminifer *Amphistegina gibbosa* (Ross and Hallock 2014). The authors reported findings as Acute Concentration (AC50) due to organism recovery after 48hr exposure and LC50 after 48hour exposure with a 24hr observational period. The 48hr AC50 and 72hr LC50 for propylene glycol was 3% and 6%v/v, respectively and 0.2% and 1%v/v for 2-butoxyethanol. Investigations on the toxicity of MC-252 and Corexit 9500A were conducted by Cohen et al (2014) using copepods *Labidocera aestiva* in 24hr and 48hr acute toxicity tests. The authors reported 48 hr LC50's in the form of total petroleum hydrocarbons of crude oil WAF treatments as 40.5 µL/L 24hr, 37.5 µL/L 48hr and CEWAFs as 190.4 µL/L 24hr and 74.3 µL/L 48hr. Corexit only treatments were reported as 7.8mg/L and 4.5mg/L for 24hr and 48hr treatments, respectively.

Almeda and associates evaluated the effect of Louisiana sweet crude (LSC), Corexit 9500A, and Corexit 9500A treated LSC mixtures exposed to lower trophic level organism. Almeda et al (2014a) reported 48hr EC50s for crude oil, dispersant, and dispersant/oil mixtures for each of the following species: *Stombidium sp* (1.73, 0.08, 1.04 µL/L), *Spirostrombidium sp* (0.99, 0.04, 0.85 µL/L), *Eutintinnus pectinis* (1.07, 0.03, 0.15 µL/L), *Favella ehrenbergii* (4.87, 0.2, 2.29 µL/L), *Gyrodinium spirale* (16.42, 0.76, 13.40 µL/L), and *Protoperidinium divergens* (13.73,0.28,5.69 µL/L). Additional 48hr tests with the same toxicants using calanoid copepods, *Arcartia tonsa, Temora turbinate,* and *Parvocalanus crassirostris* and investigated ingestion of crude oil droplets on sublethal endpoints (Almeda et al 2014b). At test concentrations of 1 µL/L SLC, 0.5 µL/L Corexit 9500A, 1 µL/L LSC/Corexit, the authors reported % reduction of observed endpoints were most toxic for dispersant/oil mixtures ranging from 45-54% egg production rates, 28-41% fecal pellet production rates, and 11-31% egg hatching. Using the same test concentrations as above, Almeda et al (2014c) conducted 72hr toxicity tests with test organism barnale nauplii (*Amphibalanus improvises*) and tornaria larvae

(*Schizocardium sp)*. The authors reported no calculable effect concentrations for *Amphibalanus improvises* and *Schizocardium sp.* as having a higher sensitivity with no reported mortalities at test concentrations. Growth inhibition concentrations (GC50) were calculated for *Schizocardium sp.* as 2.5 µL/L LSC only, 0.03 µL/L dispersant only, 0.88 µL/L dispersant/oil mixture.

Multi-species Toxicity. Median effect concentrations of three dispersants: Corexit 9500A, Dasic NS, and Gamlen OD400, and five surface washing agents: Hela saneringsvæske, Bios, Bioversal, Corext 9580, and Absorrep K212 on four pelagic species: *Skeletonema costatum, Acartia tonsa, Calanus finmarchicus, Calanus glacialis*, and one benthic species, *Corophium volutator* was evaluated (Hansen et al 2014). The 72 hr EC50 values for each species were reported as 9.3 - 2270mg/L for *S. costatum*; 48hr LC50 6.5 - 2790mg/L for *A. tonsa*, 96hr LC50 11.5 - >20435mg/L for *C. finmarchicus*, 144h LC50 9- >20435mg/L for *C. glacialis*, and 10-d LC50 140- >15347mg/L for *C. volutator*. Due to the variability of sensitivities to each product and species interaction, the authors ranked the species 1 thru 5 and concluded *A. tonsa* as the most sensitive and *C. volutator* as the least sensitive species.

Koyama et al (2014) exposed diatoms, *Chaetoceras gracilis* and *Skeletonema costatum*, amphipod *Hyale barbicornis*, and red sea bream fish embryos *Pagrus major* to physically dispersed (PDWAFs) and chemically dispersed (CEWAFs) water accommodated fractions and using heavy C oil and D1128 dispersant. The EC50 reported for PDWAF/CEWAF exposures for each species include 0.39/035 mg/L *C. gracilis*, 0.1/0.34 mg/L *S. costatum*, 0.09/0.06 mg/L *H. barbicornis*, and 0.11/0.018 mg/L *P. major*. Toxicity results were utilized in a simulation model for Tokyo Bay, Japan to determine areas of greatest risk to higher oil concentrations due to environmental factors of wind and currents, as well as, chemical dispersion. Rial et al (2014) tested 2 marine bacteria, *Phaeobacter sp.* (Ph) and *Pseudomonas sp* (Ps), one terrestrial bacteria, *Leuconostoc mesenteroides*(Lm), and embryo/larval stage sea urchin *Paracentrotus lividus* to four spill response agents Cytosol, Finasol OSR 51, Agma OSD 569 and OD4000. The authors reported the 2 marine bacteria species showed no effect to all exposure levels ranging from 0-2000 µl/L and *Leuconostoc m.* growth inhibition EC50's of 754 µL/L Finasol OSR 51 and 129 µL/L OD4000. The sea urchin embryo tests resulted in EC50 values of 34 µL/L Agma OSD 569, 26.3 µL/L CytoSol, 2.2 µL/L OD4000, and 1.2 µL/L Finasol OSR 51.

Echinoderms. Development toxicity tests using the sea urchin *Paracentrotus lividus* (ferilization) exposed to toxicant mixtures of Maya crude oil and four spill response agents (Cytosol, Agma OSD 569, OD4000, Finasol OSR 51) were conducted to determine median effect concentrations (Rial, Vazquez and Murado 2014). The authors reported 48 hr EC50 dispersant only concentrations 15.1 mL/L Cytosol, 9.8 mL/L Agma OSD 569, 2.6 mL/L OD4000, and 1.8 mL/L Finasol OSR.

Crustaceans. Larval and juvenile stage blue crab, *Callinectes sapidus*, were used to conduct 96hr acute toxicity tests using a range of Corexit 9500 concentrations (Lively and Mckenzie 2014). The larval tests were exposed to a range of 0.1-100 ppm and juvenile tests were conducted at a range of 1-1000 ppm. The authors reported no LC50 value for juvenile stage blue crab due to no significant mortality at the highest concentration and reported a 48hr LC50 value of 40.9 mg/L for larval stage tests due to no changes in mortality after 48hrs. Rodd et al. (2014) conducted 24hr acute toxicity tests using larval stage *Artemia franciscana* exposed to surface engineered carbon nanoparticles (CB) and powered activated carbon (PAC). The authors reported an LC50 value of 370 mg/L for CB exposures and 750 mg/L for PAC exposures.

Anderson, Kuhl and Anderson (2014) conducted 96hr static toxicity tests using field collected mud crabs, *Rhithropranopeus harrisii* and Louisiana sweet crude (LSC) only and Corexit 9500 treated oil test solutions. The authors did not report an LD50 for the LSC only treatment due to lack of mortality. The 96hr LD50 for the Corexit 9500A/LSC treatment was 25 g/L.

Using copepods *Labidocera aestiva*, Cohen et al. (2014) conducted acute toxicity tests with MC-252 crude oil, Corexit 9500A, and Corexit 9500A treated crude oil treatments,. The authors reported 48 hr LC50's in the form of total petroleum hydrocarbons of crude oil WAF treatments as 40.5 µL/L 24hr, 37.5 µL/L 48hr and Corexit 9500A treated crude oil as 190.4 µL/L 24hr and 74.3 µL/L 48hr. Corexit only treatments were reported as 7.8 mg/L and 4.5 mg/L for 24hr and 48hr treatments, respectively. Oil-only and oil/dispersant mixture 48-hr toxicity assays were conducted with the shrimp *Litopenaeus vannamei* exposed to Persian Gulf-Khark crude oil (CO), Iranian Naftroob crude oil dispersant (IND), Radiagreen OSD dispersant(RD), and mixtures (Delshad et al. 2014). The authors reported LC50 values of 1741 mg/L for CO, 17 mg/L for IND, 43 mg/L for RD, 631 mg/L for CO-IND, and 357 mg/L CO-RD. Acute toxicity test using standard test organism *Americamysis (=Mysidopsis)bahia* exposed to 6 crude oils and No. 2 oil physically dispersed water accommodated fractions (PDWAFs) and chemically dispersed water accommodated fractions (CEWAFs) using Corexit 9500 and 9527 at a ratio of 1:10 was evaluated by Letinski et al. (2014). The authors reported Lethal Loading values (LL50) for PDWAFs ranging from 3-533 mg/L with the lighter crude oils resulting in higher toxicity values. The results of Corexit 9500 and 9527 were combined to report LL50s ranging from 7-64 mg/L and the authors identified lower effect concentrations were due to increase exposure to hydrocarbons and not exposure to dispersant

Molluscs. Laramore et al. (2014) conducted acute and sublethal toxicity studies using the eastern oyster *Crassostrea virginica* to assess exposure effects of Macondo Canyon (MC) 252 oil alone and combined with Corexit 9500A dispersant and reported effect levels based on the following endpoints: fertilization, development, survival, and motility. The authors report negative effects of all endpoints when exposed to concentrations of total polycyclic aromatic hydrocarbon (TPAH) concentrations ranging 1.3-5 mg/L and 0.02-0.2 mg/L which were at or above measured field concentrations during the 2010 DWH event.

Fish. In order to determine presence of synergistic toxicity effects for dispersant/oil mixtures, Adams et al (2014) compared toxicity values by conducting two separate investigations. The first test exposed Atlantic herring (*Clupea arengus*) embryos to weathered Medium South American Crude (MESA) and a 1:10 dilution of Corext 9500A:MESA solution with embryo hatching at day 19 as the endpoint. The second test exposed Rainbow trout (*Oncorhynchus mykiss*) embryos to HFO-7102, Nujol mineral oil (as control), 1:20 dilution of Corexit 9500A:HFO-7102 and a range of dilutions from 1:2.5-1:20 for Corexit 9500a:Nujol oil solutions. The authors reported a 19 day EC50 for *C. arengus* as 0.15 mg/L (normal development) and 1.02mg/L (hatching success) of oil concentrations measured by flurometry. The use of a control dispersant/oil mixture resulted in no effects observed when exposed to *O. mykiss* and the authors suggest Corexit 9500A was not exposed to embryos due to binding with oil particles, 24-day LC50 values for Corexit:HFO is reported as .052%v/v and Corexit alone .00311%v/v. Mu, Jin, Ma, Lin and Wang (2014) conducted oil only WAF (Petrozuata heavy crude), oil/chemical dispersant mixtures CE-WAF (Petrozuata heavy crude and Shuangxiang No. 1), and oil/biological dispersant mixtures BE-WAF (Petrozuata heavy crude and Weipu MD-66) 25 day chronic toxicity tests using the standard test species marine medaka, *Orzias melastigma*, with deformity being primary endpoint. The authors reported EC20 values expressed as total petroleum hydrocarbons resulting in the following concentrations: 0.56 mg/L WAF, 0.78 mg/L CE-WAF, and 5.6 mg/L BE-WAF.

. **Soil and Sediment Organisms and Functions**. Oil-dispersant toxicity effects on reproductive gene expression using a standard test worm, *Caenorhabditis elegans* and test solutions mixed with crude oil obtained from the Macondo well, MC-252, and Corexit 9500A was evaluated (Polli et al 2014) investigated. Effects of increased development of apoptotic cells in reproductive regions of the worm were observed at all concentrations reported as dilutions of a stock 1:20 dispersant-oil solution ranging from 500x, 2000x, and 5000x.

Field collected marsh sediment received doses of Macondo 252 crude oil (1 mL), Corexit 9500A (0.1 mL) and oil-dispersant mixture (1 mL oil combined with 0.1 mL Corexit 9500A) in a study conducted in 2 phases marked by nitrate additions (50ppm and 100ppm) and monitored N_2O and CO_2 productions to determine changes in organic matter mineralization and denitrification (Shi and Yu 2014). The authors reported no significant changes to oil only exposures, significant inhibition of denitrification and increased mineralization when exposed to dispersants. Sediments were dosed with IFO-15 crude oil and IFO-15/Corexit 9500 dispersant mixture to investigate effects of the dispersant on natural hydrocarbon degradation in marine sediment (Macías-Zamora et al. 2014). The authors reported no significant differences between oil only and dispersant/oil mixtures for hydrocarbon degradation, aromatics were readily degraded, and PAH degradation increased linearly with oxygen availability.

Tissue Culture Toxicity. Zheng et al. (2014) conducted MMT cytotoxicity assays with skin cancer cells (B16/BL6), neuronal cells (H19-7), human astrocytoma cells (1321N1), kidney cell I (HEK-293), kidney cell II (HK2) and Corexit 9500 as the toxicant. Test concentrations ranges were 20, 40, 80, 160, and 200 ppm and exposures lasted 48hrs. The authors reported LC50 values as 16 ppm, 33 ppm, 70 ppm, 93 ppm, 95 pmm for BL16/B6, 1321N1, H19-7, HEK293, and HK-2, respectively. Sperm whale skin fibroblast cells were field collected by Wise et al. (2014) and exposed to Corexit 9500 and 9527 for 24 hours to determine cytotoxicity and clastogenicity effects. Using a clonogenic assay, the authors reported Corexit 9500 induced increased cytotoxic effects with 3% survival at the highest concentration tested of 0.1% and Corexit 9527 resulted in 26% survival at the highest concentration of 0.1%. A chromosomal aberration assay determined Corexit 9527 induced increased genotoxic effects when compared to Corexit 9500.

Comparison of Dispersants and House-hold Cleaners. Word et al. (2014) contracted two accredited EPA Laboratories to conduct 48hr and 96hr acute toxicity tests to compare LC50's among 8 household cleaning products and Corexit 9500 using standard test organisms *Americamysis (=Mysidopsis)bahia* and *Menidia beryllina*. The authors reported mean LC50 results from 48hr *A. bahia* tests for the two laboratories (lab A-Lab B) to be 28.9-35.1 ppm, 48.0-91.2 ppm, 13.3-20.7 ppm, 32.9-45.3 ppm, 35.4-67.7 ppm, 177-413 ppm, 10.7-12.4 ppm, 328-387 ppm, 40-45.3 ppm for Dawn dish soap, Restore the Earth dish soap, Palmolive dish soap, Green Works dish soap, Cascade dish detergent, Johnson's Baby shampoo, Tide Laundry detergent, Green Works all purpose cleaner, and Corexit 9500, respectively. The reported mean LC50 values from 96hr *M. beryllina* tests for the two laboratories (Lab A- Lab B) are 8.9-8.3 ppm, 26.9-21.2 ppm, 7.1-5.4 ppm, 7.8-9.9 ppm, 56.6-55.6 ppm, 38.8-42.0 ppm, 4-11.8 ppm, 386-591 ppm,35.4-110 ppm for the same respective test solutions as above.

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Table 1—Examples of chemical residues reported from marine organisms. See text for more data. Always consult cited paper to confirm actual values, units and confidence limits. See footnotes for concentration units.

Chemical	Organism Location	Geographical µg/g dry weight	Concentration,	Reference
Aluminum	Pelecypods	India	647-73019	Bhattacharya et al.
		Namibia	1.02-3.2	Dahms et al.
		So. Africa	5-1,400[c]	Greenfield et al.
Arsenic	Seaweed	Greenland	0.2-55[c]	Søndergaard et al.
	Macroalgae	Antarctica	14.8	Majer et al.
	Nemerteans	Antarctica	18.6	Majer et al.
	Amphipods	Antarctica	7.8-9.8	Majer et al.
	Isopods	Antarctica	8.4	Majer et al.
	Decapods	Antarctica	1.7-12	Majer et al.
	Pelecypods	China	0.098-0.3[c]	Tu et al.
		Korea	3-25	Hong et al.a
		So. Africa	1-13[a]	Greenfield et al.
		Adriatic Sea	4-30	Stanković et al.
		Greenland	14-16[c]	Søndergaard et al.
	Gastropods	Greenland	11-14[c]	Søndergaard et al.
		Antarctica	6-26	Majer et al.
	Echinoderms	Antarctica	5.02-6	Majer et al.
	Fish	Korea	0.64-5.4	,a et al.
		Maryland	16-66[a,c]	Dutton and Fisher
		Adriatic Sea	4.01	Rozic et al.
	Mammals	Portugal	<0.67-1.2[a]	Méndez-Fernadez et al. b
Cadmium	Green algae	Gulf of California	0.7-2.4	Hernàndez-Almaraz et al.
		Egypt	1.1-1.5	El-Din et al.
	Brown algae	Gulf of California	5.2-9.6	Hernàndez-Almaraz et al.
		Egypt	2.2-8	El-Din et al.
	Red algae	Gulf of California	5.1-6	Hernàndez-Almaraz et al.
		Egypt	1-4	El-Din et al.
	Sea grass	Adriatic Sea	0.22-1.86	Stanković et al.
	Plankton	India	4.2-21.6	Bhattacharya et al.
	Nemerteans	Antarctica	5	Majer et al.
	Polychaetes	Persian Gulf	1.2-82.2	Amoozadeh et al.
		France	0.1-0.5[c]	Dedeh et al.
	Barnacles	Persian Gulf	2.73-484	Amoozadeh et al.
	Amphipods	Antarctica	0.5-2.3	Majer et al.
	Isopods	Antarctica	1.07	Majer et al.
	Decapods	Persian Gulf	0	Dadar et al.
		Persian Gulf	0.76-5.6	Amoozadeh et al.
	Pelecypods	Persian Gulf	7.4-119,4	Amoozadeh et al.
		Namibia	0.12-1.65	Dahms et al.
		So.Africa	0.15-3.8[c]	Greenfield et al.
		France	0.1-2.7[c]	Dedeh. et al.
		Adriatic Sea	1.97-4.1	Stanković. et al.
		Greenland	2.44-3.03[c]	Søndergaard et al.
	Gastropods	Antaractica	1.76	Majer et al.
		Gulf of California	4.6-17.7	Hernàndez-Almaraz et al.
	Echinoderms	Greece	0.04-1.2	Strogylaudi et al.
		Antarctica	0.03-0.98	Majer et al.
	Fish	Sweden	0.11-0.16	Boalt et al.
		Greece	0.1-1.2	Giannakopoulou & Neofitou
		Maryland	6-46[a,c]	Dutton & Fisher
Cadmium (cont.)	Fish	Gulf of California	0.18-20.6	Ruelas-Inzuinza. et al.

		Adriatic Sea	0.01-0.016	Rozic et al.
	Mammals	Portugal	<0.07-30[a]	Méndez-Fernandez et al.b
	Birds	France	0.03-18.7	Leucie et al.
Chromium	Seaweed	Greenland	0.05-0.6[c]	Søndergaard et al.
	Seagrass	Adriatic Sea	3.2-37	Stantović et al.
	Plankton	India	21-194	Bhattacharya et al.
	Pelecypods	Namibia	0.36-1.2	Dahms et al.
		So. Africa	25-110[c]	Greenfield et al.
	Adriatic Sea	0.8-2.9	Stantović et al.	
		Greenland	1.26-2.65[c]	Søndergaard et al.
	Gastropods	Greenland	0.4-1.08[c]	Søndergaard et al.
	Echinoderms	Greece	0.3-13.7	Strogylaudi et al.
	Fish	India	0-1.55	Velusany et al.
		Greece	0-5	Giannakopoulou & Neufitou
		Maryland	4.-54[a,c]	Dutton & Fisher
		Adriatic Sea	49.1	Rozic et al.
		Kuwait	4-6	Bu-Olayan & Thomas
	Mammals	Portugal	<0.7-1.8[a]	Méndez-Fernandez et al. b
Cobalt	Plankton	India	32-111	Bhattacharya et al.
	Pelecypods	Namibia	0.36-1.2	Dahms et al.
	Decapods	Persian Gulf	0	Dadar et al.
	Mammals	Portugal	<0.07[a]	Méndez-Fernandez et al. b
Copper	Green algae	Egypt	2.5-6	El-Din et al.
		Gulf of California	1.7-3.2	Hernández-Almaraz et al.
	Brown algae	Egypt	4-4.5	El-Din et al.
		Gulf of California	1.5-4.5	Hernández-Almaraz et al.
	Red algae	Egypt	3-4.2	El-Din et al.
		Gulf of California	1.6-5.1	Hernándes-Almaraz et al.
	Sea weed	Greenland	0.1-27.5[c]	Søndergaard et al.
	Macroalgae	Antarctica	3.5	Majer et al.
	Sea grass	Adriatic Sea	0.3-16.5	Stanković et al.
	Nemerteans	Antarctic	18.5	Majer et al.
	Amphipods	Antarctic	41-58	Majer et al.
	Isopods	Antarctic	119	Majer et al.
	Decapods	Persian Gulf	0.06-4.14	Dadar et al.
	Pelecypods	Namibia	0.72-1.4	Dahms et al.
		China	0.15-0.38[c]	Tu et al.
		So. Africa	2-110[a]	Greenfield et al.
		Adriatic Sea	3.7-88.4	Stanković et al.
		Greenland	8.9-25.7[c]	Søndergaard et al.
	Gastropods	Greenland	24-75.6[c]	Søndergaard et al.
		Gulf of California	<0.07-1.3	Hernández-Almaraz et al.
		Antarctic	3.4	Majer et al.
	Echinoderms	Greece	2.1-11.8	Strongylandi et al.
		Antarctic	3.6-5.2	Majer et al.
	Fish	India	0.51-5.8	Velusang et al.
		Greece	0.1-10	Giannakopoulou &
Neafutou				
		Adriatic Sea	1.8-1.97	Kozic et al.
		Kuwait	7-10	Bu-Olayan & Thomas
	Mammals	Portugal	2.7-8.7	Méndez-Fernandez et al. b
Iron	Green algae	Gulf of California	330-372	Hernández-Almaraz et al.
	Brown algae	Gulf of California	116-135	Hernández-Almaraz et al.
	Red algae	Gulf of California	98-197	Hernández-Almaraz et al.

	Sea weed	Greenland	85-132[c]	Søndergaard et al.
	Plankton	India	1,380-51,118	Bhattacharya et al.
	Decapods	Persian Gulf	0.62-1.98	Dadar et al.
	Pelecypods	Namibia	0.89-3.3	Dahms et al.
		China	0.03-0.05[c]	Tu et al.
		So. Africa	50-190c	Greenfield et al.
		Adriatic Sea	53-719	Stanković et al.
		Greenland	267-1,292[c]	Søndergaard et al.
	Gastropods	Greenland	296-617c	Sømdergaard et al.
		Gulf of California	36.4-117.5	Hernández-Almaraz et al.
	Echinoderms	Greece	44-355	Strogylandi et al.
	Fish	India	32.1-240.5	Velusamy et al.
		Kuwait	12-14	Bu-Olayau & Thomas
	Mammals	Portugal	123-398[a]	Méndez-Fernandez et al. b
Lead	Green algae	Egypt	2-20	El-Din et al.
		Gulf of California	<0.07-3.9	Hernández-Almaraz et al.
	Brown algae	Egypt	15-18	El-Din et al.
		Gulf of California	<0.07-2.8	Hernández-Almaraz et al.
	Red algae	Egypt	1-28	El-Din et al.
		Gulf of California	<0.07-2.4	Hernández-Almaraz et al.
	Sea weed	Greenland	0.03-4.3[c]	Søndergaard et al.
	Macroalgae	Antarctica	4.5	Majer et al.
	Sea grass	Adriatic Sea	2.2-19.1	Stanković et al.
	Plankton	India	0.04-97.5	Bhattacharya et al.
	Nemerteans	Antarctica	2.6	Majer et al.
	Polychaetes	Persian Gulf	0.09-24	Amoozadel et al.
		France	2-10[c]	Dedeh et al.
	Barnacles	Persian Gulf	2.1-452	Amoozadeh et al.
	Amphipods	Antarctic	4.26-9.3	Majer et al.
	Isopods	Antarctica	8.7	Majer et al.
	Decapods	Persian Gulf	0.01-2.54	Dadar et al.
		Persian Gulf	0.01-6.9	Amoozadeh et al.
	Pelecypods	Persian Gulf	0.05-23.8	Amoozadeh et al.
		Namibia	1.55-2.35	Dahms et al.
		Portugal	0.65[a]	Freitas et al.
		China	0.005-0.015[c]	Tu et al.
		So. Africa	0.5-4[c]	Greenfield et al.
		France	2-45[c]	Dedeh et al.
		Adriatic Sea	1.15-7	Stanković
		Greenland	1.96-13[c]	Søndergaard et al.
	Gastropods	Gulf of California	<0.02-3.7	Hérnandez-Almaraz et al.
		Antarctica	5.9	Majer et al.
	Echinoderms	Antarctica	4.3-8.9	Majer et al.
	Fish	India	0-0.24	Velusamy et al.
		Gulf of California	0.04-0.06	Ruelas-Inzunza et al.
		Adriatic Sea	0.1-0.65	Rozic et al.
		Kuwait	2-4	Buo-Olayan & Thomas
		Portugal	<0.07[a]	Méndez-Fernandez et al. b
	Birds	France	0.01-0.38	Lucia et al.
Manganese	Plankton	India	122-1,066	Bhattacharya et al.
	Polychaetes	Greece	0.5-13	Kalantzi et al.
	Anthropods	Greece	0.4-2.1	Kalantzi et al.
	Molluscans	Greece	0.1-0.4	Kalantzi
Manganese (cont.)	Pelecypods	Adriatic Sea	2-13	Stanković et al.
		Namibia	0.64-1.98	Dahms et al.
		So. Africa	2-64[c]	Greenfield et al.
	Echinoderms	Greece	0.6-2.4	Kalanzi et al.
	Fish	India	1.17-7.75	Velusamy et al.
	Mammals	Portugal	<2.5-3.4[a]	Méndez-Fernandez et al. b

Mercury	Phytoplankton	Brazil	0.0017-0.00175	Seixas et al.
		Brazil	0.0013[a]	Silva-Filho et al.
	Seaweed	Greenland	0.03[c]	Søndergaard et al.
	Zooplankton	Brazil	0.025[a]	Silva-Filho et al.
	Polychaetes	France	0.3-2[c]	Dedeh et al.
		Florida	0.013[a]	Thera & Rumbold
	Isopods	Florida	0.015[a]	Thera & Rumbold
	Decapods	Florida	0.014-0.098[a]	Thera & Rumbold
		Mediterranean Sea		1.8 Cresson et al.
	Pelecypods	France	0.1-2	Dedeh et al.
		Greenland	0.1-0.11[c]	Søndergaard et al.
	Gastropods	China	27.7	Ho & Leung
		Greenland	0.04-0.05[c]	Søndergaard et al.
		Florida	0.008-0.02[a]	Thera & Rumbold
	Cephalopods	Florida	0.017[a]	Thera & Rumbold
	Echinoderms	Florida	0.004-0.03[a]	Thera & Rumbold
	Fish	Brazil	0.08-0.34	Seixas et al.
		Florida	0.00037	Rumbold et al.
		Florida	0.03-0.48	Huge et al.
		Florida	0.04-2.84[a]	Thera & Rumbld
		Maryland	4-58[a,c]	Dutton & Fisher
		India	0.01-0.23	Velusamy et al.
		Sweden	0.52-1.63	Boalt et al.
		Adriatic Sea	0.07-4.4	Horvat et al.
		Mediterranean Sea		1.3-7.13 Cresson et al.
		Kuwait	1-2	Bu-Olaguo & Thomas
	Mammals	Brazil	0.046-1.8	Seixas
		Portugal	1.6-31[a]	Méndez-Fernandez et al. b
	Birds	France	0.15	Lucia et al.
Nickel	Seaweed	Greenland	0.08-2.5[c]	Søndergaard et al.
	Macroalgae	Antarctica	0.85	Majer et al.
	Nemerteans	Antarctica	0.48	Majer et al.
	Amphipods	Antarctica	0.86-1.8	Majer et al.
	Isopods	Antarctica	0.78	Majer et al.
	Decapods	Persian Gulf	0.29-0.31	Dadar et al.
	Pelecypods	Namibia	0.35-1.95	Dahms et al.
		China	0.02-0.04[c]	Tu et al.
		So. Africa	0-15[c]	Greenfield et al.
		Adriatic Sea	0.8-5	Stanković et al.
		Greenland	2.2-9.4[c]	Søndergaard et al.
	Gastropods	Greenland	2-3.08[c]	Søndergaard et al.
		So. Africa	0.37	Majer et al.
	Echinoderms	Greece	2.4-26.7	Strogyloudi et al.
		So. Africa	0.6-1.16	Majer et al.
	Fish	Kuwait	6-8	Bu-Olayou & Thomas
	Mammals	Portugal	<0.2-0.8[a]	Méndez-Fernandez et al. b
Selenium	Fish	Adriatic Sea	0.87-1.78	Rozic et al.
	Mammals	Portugal	2.7-16.9[a]	Méndez-Fernandez et al. b
Silver	Seaweed	Greenland	0.01-0.19[c]	Søndergaard et al.
	Pelecypods	Greenland	0.04-0.1[c]	Søndergaard et al,
	Gastropods	Greenland	1.08-1.35[c]	Søndergaard et al.
	Mammals	Portugal	<0.07[a]	Méndez-Fernandez et al. b
	Birds	France	0.01-1.7	Lucia et al.
Strontium	Pelecypods	So. Africa	5-120[c]	Greenfield et al.
Tin	Gastropods	R. Cabo Verde	0.005-0.037	Lopes-dos-Santos et al.

	Fish	Brazil	0.032-0.33	dos Santos et al.
Vanadium	Polychaetes	Greece	3.4-53.8	Kalantzi et al.
	Arthropods	Greece	1.7-6.6	Kalantzi et al.
	Molluscans	Greece	0.4-3.3	Kalantzi et al.
	Pelecypods	China	0.015-0.03[c]	Tu et al.
	Echinoderms	Greece	1.3-4.5	Kalantzi et al.
	Mammals	Portugal	<0.1[a]	Méndez-Fernandez et al. b
Zinc	Green algae	Egypt	8-40	El-Din et al.
		Gulf of California	16-20.8	Hernández-Almaraz et al.
	Brown algae	Egypt	17-45	El-Ding
		Gulf of California	15.3-23	Hernández-Almaraz et al.
	Red algae	Egypt	10-43	El-Din
		Gulf of California	9.1-18.7	Hernández-Almaraz et al.
	Seaweed	Greenland	0.05-40[c]	Sandergaard et al.
	Macroalgae	Antarctica	21.7	Majer et al.
	Plankton	India	725-1,670	Bhattacharya et al.
	Nemerteans	Antarctica	158	Majer et al.
	Amphipods	Antarctica	52	Majer et al.
	Isopods	Antarctica	53	Majer et al.
	Decapods	Persian Gulf	0.34-0.38	Dadar et al.
	Pelecypods	Namibia	1.37-2.92	Dahms et al.
		So. Africa	50-220[c]	Greenfield et al.
		Adriatic Sea	2-245	Stanković et al.
		Greenland	8.9-30.1[c]	Søndergaard et al.
	Gastropods	Greenland	24-75[c]	Søndergaard et al.
		Gulf of California	24.7-30	Hernández-Almaraz et al.
		Antarctica	23.7	Majer et al.
	Echinoderms	Antarctica	58-353	Majer et al.
		Greece	19-1,024	Strogylocede et al.
	Fish	India	12.77-60.75	Velusamy et al.
		Adriatic Sea	23.5-136	Kozic et al.
		Kuwait	10-25	Bu-Olayou & Thomas
		Greece	5-90	Giannakopoulou & Neofitou
	Mammals	Portugal	18.8-53[a]	Méndez-Fernandez et al. b
Biphenols	Phytoplankton	Baltic Sea	0.0057-0.24	Staniszewska et al.
	Zooplankton	Baltic Sea	0-0.77	Staniszewska et al.
	Pelycypods	Baltic Sea	0-0.197	Staniszewska et al.
Chlorinated Paraffins	Decapods	China	0.003-1.05[a]	Ma et al. a
	Pelecypods	China	4.77-32.7	Ma et al. b
		China	0.0037-2.2[a]	Ma et al. a
	Gastropods	China	0.0026-1.05[a]	Ma et al. a
	Fish	China	0.003-2.9[a]	Ma et al. a
Chlorodane	Fish	California	0.008-2.8	Lyons et al.
∑DDT	Pelecypods	Korea	0.0001-0.003	Choi,Yang, Hong and Chin
		Massachusetts	0.78[b]	Subedi, at al.
	Fish	China	0.008-0.04[b]	Hao et al.
		China	0097-5.3[b]	Sun et al.
		Poland	0.006-0.035[a]	Waszak et al.
		Portugal	0.0007-0.0024	Waszak et al.
		California	0.004-2.4[b]	Lyons et al.
	Mammals	Brazil	0.000003-0.005[b]	Santos-Neto et al.
Heptachlor	Fish	Poland	0.00006-0.0001[a]	Waszak et al.
		Portugal	0.00002-0.00004[a]	Waszak et al.

∑HCBD	Fish	Poland	0.00008-0.0005[a]	Waszak et al.
		Portugal	0.00002-0.0001[a]	Waszak et al.
ΣHCHe	Pelecypods	Korea	0.00009-0.0003	Choi,Yang, Hong and Chin
	Mammals	Brazil	0.000005-0.00016[b]	Santos-Neto et al.
Methyl siloxanes	Fish	China	5.1[a]	Hong et al.b
Mirex	Decapods	China	0.00064-0.0015[a]	Peng et al.
	Pelecypods	China	0.001-0.02[a]	Peng et al.
	Fish	China	0.006-0.098[a]	Peng et al.
	Mammals	Brazil	0.0002-0.024[b]	Santos-Neto et al.
	Birds	China	0.3-2.3[a]	Peng et al.
PAHs	Brown algae	Aegean Sea	0.75-5	Apostolopoulou et al.
	Corals	Taiwan	`0.14-1.7	Ko, Chang and Cheng
PAHs cont.	Pelecypods	Turkey	0.4-589	Balcioğlu et al.
	Starfish	Korea	1.2-1.49	Kim, Hong, Kim and Yang
	Fish	Chukchi Sea	0.65-5.2	Harvey et al.
		China	0.01-0.94	Li, Lu, Huang, Yang and Ran
		Spain	0.003-0.04	León et al.
PBDEs	Pelecypods	Massachusetts	277[b]	Subedi et al.
	California	0.001-0.1	Dodder et al.	
	Fish	China	0.0029-0.0078[b]	Hao et al.
		China	0.002-0.12[b]	Sun et al.
		Poland	0.0002-0.002[a]	Waszak et al.
		Portugal	0.0002-0.0007[a]	Waszak et al.
		Germany	0.002-0.0084[a]	Sühring et al.
		Canada	0.004-0.077[a]	Sühring et al.
	Mammals	China	0.0013-0.049[b]	Zhu et al.
		China	0.0027-0.94[b]	Ko, We and Chou
∑PCBs	Pelecypods	Korea	0.001-0.0035	Choi,Yang, Hong and Chin
		Massachusetts	182-942[b]	Sibedi et al.
	Fish	China	0.014-0.048[b]	Hao et al.
		China	0.006-0.2[b]	Sun et al.
		Poland	0.0049-0.016[a]	Waszak et al.
		Portugal	0.0027-0.0087[a]	Waszak et al.
		California	0.22-48.2[b]	Lyons et al.
	Polar Bear	Arctic Ocean	1-8[b]	Binnington & Wania
	Mar. Mammals	Portugal	2-61.2	Méndez-Fernandez et al. a
		Brazil	0.00002-0.017[b]	Santos-Neto et al.
		Arctic Ocean	0.1-1.2[b]	Binnington & Wania
		Canada	0.001-0.065[b]	Binnington & Wania
Perfluorinated compounds	Pelecypods	California	0-0.009	Dodder et al.
Pharmaceuticals	Pelecypods	California	0.001-0.15	Dodder et al.
		UK	0.004-0.029[c]	McEnuff et al.

a=Wet weight
b=Experimental study
c=Lipid weight
d=Polycyclic aromatic hydrocarbons
e=Polybrominated diphenyl ethers