CHAPTER SEVEN: CHEMICAL EFFECTS—WATER DISCHARGE FACILITIES

Introduction

Disposal of various waste materials into rivers, estuaries, and marine waters is not a modern phenomenon; this practice has been used as a preferred disposal option virtually since the beginning of human civilization (Ludwig and Gould 1988; Islam and Tanaka 2004). Nevertheless, when the full spectrum of emissions from land-based activities is taken into account, the use of coastal waters as a repository for anthropogenic waste has not previously been practiced on as large or intense a global scale as in recent decades (Williams 1996). In the United States, growing human population densities in coastal communities have manifested a demonstrably adverse effect on aquatic resources. The scientific literature is replete with evidence of inorganic and organic pollutant accumulation in coastal waters from anthropogenic effluents (e.g., Ragsdale and Thorhaug 1980; Tessier et al. 1984; Phelps et al. 1985; Long E et al. 1995; Pastor et al. 1996; Smith et al. 1996; Chapman and Wang 2001; Hare et al. 2001; O'Connor 2002; Robinet and Fenteun 2002; Wurl and Obbard 2004). The federal Clean Water Act (CWA), enacted in 1972 to address many of these issues, eliminated certain types of disposal activities and otherwise induced improvements to the nation's surface water quality. Nonetheless, "despite reductions in pollution from municipal and industrial point sources more than one-third of the river miles, lake acres, and estuary square miles suffer [*sic*] some degree of impairment" (Ribaudo et al. 1999). To the extent that it may alter natural processes and natural resource communities, unabated degradation of the aquatic environment caused by a wide spectrum of human activities poses consequences for fishery resources and their habitats.

Contaminants enter our waterways through two generic vectors: point and nonpoint sources. Pollutants of nonpoint source origins tend to enter aquatic systems as relatively diffuse contaminant streams primarily from atmospheric and terrestrial sources (see Coastal Development chapter of this report for discussions on nonpoint source pollution). In contrast, point source pollutants generally are introduced via some type of pipe, culvert, or similar outfall structure. These discharge facilities typically are associated with domestic or industrial activities, or in conjunction with collected runoff from roadways and other developed portions of the coastal landscape. Waste streams from sewage treatment facilities and watershed runoff in many urbanized portions of the northeastern United States are first intermingled and then subsequently released into aquatic habitats via combined sewer overflows (CSOs). Such point discharges collectively introduce a cocktail of inorganic and organic contaminants into aquatic habitats, where they may become bioavailable to living marine resources.

While all pollutants can become toxic at high enough levels, there are a number of compounds that are toxic even at relatively low levels. The US Environmental Protection Agency (US EPA) has identified and designated more than 126 analytes as "priority pollutants." According to the US EPA, "priority pollutants" of particular concern for aquatic systems include: (1) dichlorodiphenyl trichloroethane (DDT) and its metabolites; (2) chlorinated pesticides other than DDT (e.g., chlordane and dieldrin); (3) polychlorinated biphenyl (PCB) congeners; (4) metals (e.g., cadmium, copper, chromium, lead, mercury); (5) polycyclic aromatic hydrocarbons (PAHs); (6) dissolved gases (e.g., chlorine and ammonium); (7) anions (e.g., cyanides, fluorides, and sulfides); and (8) acids and alkalis (Kennish 1998; USEPA 2003a). While acute exposure to these substances produce adverse effects of aquatic biota and habitats, chronic exposure to low concentrations probably is a more significant issue for fish population structure and may result in multiple substances acting in “an additive, synergistic or antagonistic manner” that may render impacts relatively difficult to discern (Thurberg and Gould 2005).

Determining the eventual fate and effect of naturally occurring and synthetic contaminants in coastal environments and biota is a highly dynamic proposition that requires interdisciplinary evaluation. It is essential that all processes sensitive to pollutants be identified and that investigators realize that the resulting adverse effects may be manifested at the biochemical level in organisms (Luoma 1996) in a manner particular to the species or life stage exposed. Pollutant exposure can inhibit: (1) basic detoxification mechanisms, like production of metallothioneins or antioxidant enzymes; (2) the ability to resist diseases; (3) the ability of individuals or populations to counteract pollutant-induced metabolic stress; (4) reproductive processes including gamete development and embryonic viability; (5) growth and successful development through early life stages; (6) normal processes including feeding rate, respiration, osmoregulation; and (7) overall Darwinian fitness (Capuzzo and Sassner 1977; Widdows et al. 1990; Nelson et al. 1991; Stiles et al. 1991; Luoma 1996; Thurberg and Gould 2005).

The nature and extent of a pollutant's dispersal in our waterways are collectively dependent on a variety of factors including site-specific ecological conditions, the physical state in which the contaminant is introduced into the aquatic environment, and the inherent chemical properties of the substance in question. Soluble or miscible substances typically enter waterways in an aqueous phase and eventually become adsorbed onto organic and inorganic particles (Wu et al. 2005); however, contaminants may enter aquatic systems as either particle-borne suspensions or as solutes (Bishop 1984; Turner and Millward 2002). Dilution and settling out from such effluent streams initially are dictated by physical factors (e.g., the presence of significant currents or perhaps a strong thermocline or pycnocline) which predominantly influence the spatial extent of contaminant dispersal. In particular, turbulent mixing, or diffusion, disperses contaminant patches in coastal waters resulting in larger, comparatively diluted contaminant distributions further away from the initial point source (Bishop 1984). Biological activity and geochemical processes subsequently intercede and typically result in contaminant partitioning between the aqueous and particulate phases (Turner and Millward 2002).

While physical dispersion, biological activity, and other ecological factors clearly have important roles regarding the distribution of contaminants in aquatic habitats, contaminant partitioning is largely governed by certain ambient environmental conditions, notably salinity, pH, and the physical nature of local sediments (Turekian 1978; McElroy et al. 1989; Turner and Millward 2002; Leppard and Droppo 2003; Wu et al. 2005). Highly reactive suspended particles typically serve as important carriers of aquatic contaminants and largely are responsible for their bioavailability, transport, and ecological fate as they become dispersed in receiving waters (Turner and Millward 2002). In addition, hyporheic (i.e., the saturated zone under a river or stream, comprising substrate with the interstices filled with water) exchange between overlying water and groundwater can alter salinity, dissolved oxygen concentration, and other water chemistry aspects in ways that can influence the affinity of local sediment types for particular contaminants or otherwise affect contaminant behavior (Ren and Packman 2002).

Amendments to the CWA include important provisions to address acute or chronic water pollution emanating from discharge pipes and outfalls under the National Pollutant Discharge Elimination System (NPDES) program. Until the late 1980s, the NPDES program traditionally focused efforts on controlling industrial and municipal sewage discharges but has since expanded its purview to include storm water management (USEPA 1996). While the NPDES program has led to ecological improvements in waters of the United States, point sources continue to introduce pollutants into the aquatic environment, albeit at reduced levels. Nonetheless, studies demonstrate that particle-associated contaminants collected in coastal depositional areas are preserved in chronological strata or horizons (Huntley et al. 1995; Chillrud et al. 2003). Consequently, historically deposited contaminants may be encountered when installing new outfalls or coastal infrastructure, especially near urbanized areas. Regardless of whether these pollutants were deposited recently or decades ago, dredging incidental to construction and related activities that enhance their potential biological availability can have adverse ecological implications.

The environmental dynamics of point source wastes are complex and involve a variety of physical, chemical, and biological processes simultaneously acting on the introduced suite of contaminants and their surrounding habitat. Because of the many competing variables involved, it is difficult to predict the ultimate fate and effects of anthropogenic wastes with great precision; however, local habitat characteristics in combination with the relative solubility, degree of hydrophobicity (i.e., tending to repel and not absorb water), and chemical reactivity of the introduced substances are important determining factors at the most basic level of analysis.

To minimize redundancy, all recommended conservation measures and best management practices for sewage discharge facilities, industrial discharge facilities, and combined sewer overflows have been included at the end of this chapter.

Sewage Discharge Facilities

Introduction

Sewage treatment plants introduce a host of contaminants into our waterways primarily through discharge of fluid effluents comprising a mixture of processed "black water" (sewage) and "gray water" (all other domestic and industrial wastewater). Such municipal effluents begin as a complex mixture of human waste, suspended solids, debris, and a variety of chemicals collectively derived from domestic and industrial sources. These contaminants include an array of suspended and dissolved substances, representing both inorganic and organic chemical species (Grady et al. 1998; Epstein 2002). These substances potentially include the full spectrum of EPA priority pollutants mentioned previously and many other contaminants of anthropogenic origin. However, the five constituents that are usually the most important in determining the type of treatment that will be required are: (1) organic content (usually measured as volatile solids); (2) nutrients; (3) pathogens; (4) metals; and (5) toxic organic chemicals (USEPA 1984).

Coastal communities rely on municipal wastewater treatment to contend with potential human health issues related to sewage and also to protect surface and groundwater quality. Municipal processing facilities typically receive raw wastewater from both domestic and industrial sources, and are designed to produce a liquid effluent of suitable quality that can be returned to natural surface waters without endangering humans or producing adverse aquatic effects (Grady et al. 1998; Epstein 2002). As it is currently practiced in the United States, wastewater treatment entails subjecting domestic and industrial effluents to a series of physical, chemical, or even biological processes designed to address or manipulate different aspects of contaminant mitigation. For both logistical and economic reasons, not all municipalities expend the same level of effort removing contaminants from their wastewater before returning it to a receiving aquatic habitat. The following discussion summarizes the different levels that municipal wastewater treatment and resulting water quality benefits derived from them.

Primary treatment, also known as "screen and grit," is only marginally effective at addressing sewage contaminants and simply entails bulk removal of "settleable" solids from the wastewater by sedimentation and filtration. Sometimes total suspended solids are further reduced in the initial effluent treatment phase by implementing another level of primary treatment, which entails using chemicals to induce coagulation and flocculation of smaller particles (Parnell 2003).

The resulting bio-solids must be disposed, and their final disposition could entail composting with subsequent use in agricultural applications, placement in a landfill, disposal at sea, or even incineration (Werther and Ogada 1999). Removal and appropriate disposal of sewage present in a solid phase are important steps, if elementary, in addressing human health and aesthetic issues surrounding sewage management because doing so removes visible substances that otherwise would accumulate in the aquatic environment at or near the discharge point. Unfortunately, primary treatment of municipal wastewater alone often fails to meet overall environmental goals of supporting important water-dependent uses like fishery resource production and recreational uses featuring primary contact with the water. As a consequence, coastal communities in the northeastern region process their wastewater through one or more additional treatment levels beyond bulk solids removal to address the environmental challenges of their sewage effluents more effectively.

Following bulk sludge removal, sewage treatment plants typically pass the highly organically-enriched water emerging from primary treatment through a second process that is intended to address biological oxygen demand (BOD), an indirect measure of the concentration of biologically degradable material present in organic wastes that reflects the amount of oxygen necessary to break down those substances in a set time interval. Such secondary treatment, which is required for all municipal wastewater treatment in the United States, involves removal of much of the remaining organic material by introducing aerobic microorganisms under oxygen-enriched conditions (Parnell 2003). The resulting microbial action breaks organic substrates into progressively simpler compounds, with the final waste components predominantly released as carbon dioxide. The bacteria subsequently are removed by chlorination before the secondarily-treated effluent is released into local surface waters or the secondarily treated wastewater is directed to another part of the sewage treatment plant for additional processing. Where practiced, such effluent-polishing or advanced treatment measures use any of several techniques to remove inorganic nitrogenous or phosphorous salts to reduce the final effluent's potential to cause excessive nutrient enrichment of the receiving waters (Epstein 2002; Parnell 2003).

Because of the large expense of tertiary sewage treatment, the public sector does not implement it as a uniform municipal wastewater treatment policy. Consequently, while secondary treatment is the standard operating procedure for municipal wastewater treatment in the northeastern United States, natural resource managers cannot assume that advanced, tertiary treatment is available to meet desired environmental goals. Recent point source management policy decisions by Boston, MA, area communities are a case in point. Rather than implementing more costly advanced treatment during system upgrades, these communities chose to address local municipal wastewater challenges by implementing primary and secondary treatment combined with source reduction of certain contaminants and offshore diversion of outfalls to encourage enhanced effluent dilution (Moore et al. 2005). Despite the added expense of implementing them, both secondary and advanced treatment processes are important potential habitat protection measures, particularly because they mitigate oxygen depletion events, eutrophication, and related phenomena that can result in adverse ecological conditions.

Release of nutrients and eutrophication

Particularly under lesser levels of treatment, municipal sewage facilities discharge large volumes of nutrient-enriched effluent. While some level of readily available nutrients are essential to sustain healthy aquatic habitats and ecological productivity, excess concentrations result in eutrophication of coastal habitats. Elevated nitrogen and phosphorous concentrations in municipal wastewater effluents can cause pervasive ecological responses including: exaggeration of phytoplankton and macroalgal populations; initiation of harmful algal blooms (Anderson et al. 2002); adverse affects on the physiology, growth, and survival of certain ecologically important aquatic plants (Touchette and Burkholder 2000); reduction of water transparency with accompanying adverse effects to submerged and emergent vascular plants or other disruptions to the normal ecological balance among vascular plants and algae (Levinton 1982; Cloern 2001); hypoxic or anoxic events that may cause significant fish and invertebrate mortalities; disturbances to normal denitrification processes; and concomitant decrease in local populations of fishery resources and forage species (USEPA 1994). Sewage outfalls also may become an attraction nuisance in that they may at least initially attract fish around the point of discharge until hypoxia, toxin production, and algal bloom development render the aquatic area less productive (Islam and Tanaka 2004). Collectively, adverse chemical effects may be especially significant to aquatic resources in temperate regions because strong thermoclines and persistent ice cover restrict vertical mixing and exacerbate deteriorating habitat conditions at depth.

For additional information on the mechanisms involved in denitrification of organic and inorganic compounds, Korom's (1992) review of denitrification in natural aquifers is a concise and informative compilation of heterotrophic and autotrophic denitrifiers.

Release of contaminants

Municipal treatment facilities discharge large volumes of effluent into the aquatic environment. The waste stream typically contains a complex mixture of domestic and industrial wastes that contain predominantly natural and synthetic organic substances, metals, and trace elements, as well as pathogens (Islam and Tanaka 2004). Similarly, introductions of certain pharmaceuticals via municipal wastewater discharges have become causes for concern because of their potential to act as endocrine disruptors in fish and other aquatic resources. Residence time of the different contaminant classes in aquatic environments is an important habitat management consideration. Some of these substances, such as volatile organic compounds, may have a relatively short residence time in the system and other, more persistent substances, such as synthetic organometallic compounds, may linger for decades after becoming associated with the substrate or concentrated in local biota. Such pollution has been associated with mortality, malformation, abnormal chromosome division, and higher frequencies of mitotic abnormality in adult fish from polluted areas compared with those from less polluted regions of the northwest Atlantic Ocean (Longwell et al. 1992).

Increased concentrations of the various contaminant classes associated with municipal wastewater can be highly ecologically significant. For instance, exposure to contaminants within these categories have been correlated with deleterious effects on aquatic life including larval deformities in haddock (*Melanogrammus aeglefinus*) (Bodammer 1981), reduced hatching success and increased larval mortality in winter flounder (*Pseudopleuronectes americanus*) (e.g., Klein-MacPhee et al. 1984; Nelson et al. 1991), skeletal deformities in Atlantic cod (*Gadus morhua*) (Lang and Dethlefsen 1987), inhibited gamete production and maturation in sea scallops (*Placopecten magellanicus*) (Gould et al. 1988), and reproductive impairment in Atlantic cod (Thurberg and Gould 2005).

Laboratory experiments with pesticides have shown a positive relationship between malformation and survival of embryos and larvae of Atlantic cod and concentration of DDT and its breakdown product dichlorodiphenyl dichloroethylene (DDE) (Dethlefsen 1976). The proportion of fin erosion in winter flounder collected on contaminated sediments was found to be greater in fish sampled with higher concentrations of PCB in muscle, liver, and brain tissues than in fish collected in reference sites (Sherwood 1982). Studies conducted in the harbor of New Haven, CT, found high occurrences of liver lesions, blood cell abnormalities, liver DNA damage, and liver neoplasms among winter flounder with high concentrations of organic compounds, metals, and PCB in their gonads (Gronlund et al. 1991). Such pollution also has been associated with mortality, malformation, abnormal chromosome division, and higher frequencies of mitotic abnormality in adult fish from polluted areas compared with those from less polluted regions of the northwest Atlantic Ocean (Longwell et al. 1992). Observed effects of fish exposed to PAH include decrease in growth, cardiac disfunction, lesions and tumors of the skin and liver, cataracts, damage to immune systems, estrogenic effects, bioaccumulation, bioconcentration, trophic transfer, and biochemical changes (Logan 2007).

For almost a century, sewage sludge (the solids extracted from raw wastewater during sewage treatment) was disposed of at sea. In the northeastern United States, a number of designated offshore sewage sludge dumpsites existed, including one in Boston Harbor, MA, and sites in the New York Bight and the Mid-Atlantic Bight (Barr and Wilk 1994). Not surprisingly, sediments sampled in the vicinity of sewage sludge dumpsites have contained higher levels of contaminants (e.g., PCB, PAH, chlorinated pesticides, and metals) than in control sites (Barr and Wilk 1994). Sewage sludge has been demonstrated to have adverse effects on aquatic organisms. For example, early life stages of Atlantic herring (*Clupea harengus*) have shown a series of developmental abnormalities, including premature hatching accompanied by reduced viability of emerging fry; poor larval survival; smothering or incapacitation of larvae by particle flocs; and fin damage (Urho 1989; Costello and Gamble 1992). The Ocean Dumping Ban Act of 1988 prohibited sewage sludge and industrial wastes from being dumped at sea after December 31, 1991. This law is an amendment to the Marine Protection, Research, and Sanctuaries Act of 1972, which regulates the dumping of wastes into ocean waters.

In addition to these diverse contaminant classes, wastewater facilities also discharge a host of synthetic hormones or other substances that could disrupt normal endocrine function in aquatic vertebrates, as well as introduce zoonotic viruses, bacteria, and fungi that may be present in raw human sewage. These chemicals act as "environmental hormones" that may mimic the function of the sex hormones (Thurberg and Gould 2005). Adverse effects include reduced or altered reproductive functions, which could result in population-level impacts. Metals, PAHs, and other contaminants have been implicated in disrupting endocrine secretions of marine organisms (Brodeur et al. 1997; Thurberg and Gould 2005). However, the long-term effect of endocrine-disrupting substances on aquatic life is not well understood and demands serious attention by the scientific and resource policy communities. Refer to the Endocrine Disruptors subsection of this chapter for a broader discussion on this topic. Metals such as mercury are also capable of moving upward through trophic levels and can accumulate in fish (i.e., bioaccumulation) at levels which may cause health problems in human consumers.

While modern sewage treatment facilities undeniably reduce the noxious materials present in raw wastewater and some substances typical of processed effluents have their own inherent toxic effects, it also is important to recognize that secondary and advanced treatment can alter the chemistry of ordinarily benign materials in ways that initiate or enhance their toxicity. In particular, normally nonhazardous organic compounds present in wastewater potentially can be rendered toxic when raw municipal effluent is chlorinated in the sewage treatment process (NRC 1980; Epstein 2002). Other contaminants may become toxic to humans or many different aquatic resource taxa when these substances are methylated (addition of a $-CH_4$ group) or otherwise after having been chemically transformed into a harmful, biologically available molecular form.

The behavior and effects of trace chemicals in aquatic systems largely depend on the speciation and physical state of the pollutants in question. A detailed description concerning contaminant partitioning and bioavailability is beyond the scope of this technical discussion.

However, Gustafsson and Gschwend (1997) offer an excellent review of the matter in terms of how dissolved, colloidal and settling particle phases affect trace chemical fates and cycling in aquatic environments. While the observations provided by these Massachusetts Institute of Technology researchers pertain specifically to cycling of compounds in natural waters, the generic properties they discuss also would apply in the context of substances in treated wastewater since they are subject to the same physical and chemical forces. In addition, Tchobanoglous et al. (2002) may be consulted for an authoritative technical review of the environmental engineering aspects of wastewater treatment.

Exposure to potentially mutagenic or teratogenic pollutants and the resulting declines in viability at any life stage reduce the likelihood of maturation and eventual recruitment to adulthood or a targeted fishery. Literature on the aqueous and sedimentary geochemistry and physiological effects of contaminants on aquatic biota should be consulted to determine the fate of persistent compounds in local sediments and associated pore-water and the extent of acute or chronic toxic effects on affected aquatic biota (Varanasi 1989; Allen 1996; Langmuir 1996; Stumm and Morgan 1996; Tessier and Turner 1996; Paquin et al. 2003).

Alteration of water alkalinity

Municipal sewage effluent that does not meet water quality standards can alter the alkalinity of riverine receiving waters. However, freshwater and low-salinity waters with low buffering capacity are more susceptible to acidification than are marine waters. Acidification of riverine habitats has been linked to the disruption of reproduction, development, and growth of anadromous fish (USFWS and NMFS 1999; Moring 2005). For example, osmoregulatory problems in Atlantic salmon (*Salmo salar*) smolts have been related to habitats with low pH (Staurnes et al. 1996). In estuarine waters, low pH has been shown to cause cellular changes in the muscle tissues of Atlantic herring which may lead to a reduction in swimming ability (Bahgat et al. 1989). However, all municipal sewage facilities are required to obtain water quality permits through the US EPA's NPDES program and must meet established pH standards for receiving waters. Acid precipitation from atmospheric sources is of concern in the northeastern United States. Refer to the Global Effects and Other Impacts chapter for more information regarding acid precipitation.

Impacts to submerged aquatic vegetation

Submerged aquatic vegetation (SAV) requires relatively clear water in order to allow adequate light transmittance for metabolism and growth. Sewage effluent containing high concentrations of nutrients can lead to severely eutrophic conditions. The resulting depression of dissolved oxygen and diminished light transmittance through the water may result in local reduction or even extirpation of SAV beds that are present before habitat conditions become too degraded to support them (Goldsborough 1997). Examples of large scale SAV declines have been seen throughout the eastern coastal states, most notably in Chesapeake Bay, MD/VA, where overall abundance has been reduced by 90% during the 1960s and 1970s (Goldsborough 1997). Although a modest recovery of the historic SAV distribution has been seen in Chesapeake Bay over the past few decades, reduced light penetration in the water column from nutrient enrichment and sedimentation continues to impede substantial restoration. Primary sources of nutrients into Chesapeake Bay include fertilizers from farms, sewage treatment plant effluent, and acid rain (Goldsborough 1997). Short and Burdick (1996) correlated eelgrass losses in Waquoit Bay, MA, with anthropogenic nutrient loading primarily as a result of increased number of septic systems from housing developments in the watershed.

Eutrophication can alter the physical structure of SAV by decreasing the shoot density and blade stature, decreasing the size and depths of beds, and stimulating excessive growth of macroalgae (Short et al. 1993). An epidemic of an eelgrass wasting disease wiped out most eelgrass beds along the east coast during the 1930s, and although some of the historic distribution of eelgrass has recovered, eutrophication may increase the susceptibility of eelgrass to this disease (Deegan and Buchsbaum 2005).

Reduced dissolved oxygen

The decline and loss of fish populations and habitats because of low dissolved oxygen concentrations is "one of the most severe problems associated with eutrophication in coastal waters" (Deegan and Buchsbaum 2005). The effect of chronic, diurnally fluctuating levels of dissolved oxygen has been shown to reduce the growth of young-of-the-year winter flounder (Bejda et al. 1992). High nutrient loads into aquatic habitats can cause hypoxic or anoxic conditions, resulting in fish kills in rivers and estuaries (USEPA 2003b; Deegan and Buchsbaum 2005) and potentially altering long-term community dynamics (NRC 2000; Castro et al. 2003). Highly eutrophic conditions have been reported in a number of estuarine and coastal systems in the northeastern United States, including Boston Harbor, Long Island Sound, NY/CT, and Chesapeake Bay (Bricker et al. 1999). For the southern portions of the northeast coast (i.e., Narragansett Bay, RI, to Chesapeake Bay), O'Reilly (1994) described chronic hypoxia (low dissolved oxygen) as a result of coastal eutrophication in several systems. This author reported episodic, low dissolved oxygen conditions in some of the northern portions of the northeast coast, such as in Boston Bay/Charles River and the freshwater portion of the Merrimack River, MA/NH (O'Reilly 1994). Areas particularly vulnerable to hypoxia are those that have restricted water circulation, such as coastal ponds, subtidal basins, and salt marsh creeks (Deegan and Buchsbaum 2005). While any system can become overwhelmed by unabated nutrient inputs or nutrient enrichment, the effects of these generic types of pollution when experienced in temperate regions may be especially significant in the summer. This is primarily a result of stratification of the water column and higher water temperatures and metabolic rates during summer months (Deegan and Buchsbaum 2005).

Siltation, sedimentation, and turbidity

Municipal sewage outfalls, especially those that release untreated effluent from storm drains, can release suspended sediments into the water column and the adjacent benthic habitat. Increased suspended particles within aquatic habitats can cause elevated turbidity levels, reduced light transmittance, and increased sedimentation of benthic habitat which may lead to the loss of SAV, shellfish beds, and other productive fishery habitats. Other affects from elevated suspended particles include respiration disruption of fishes, reduction in filtering efficiencies and respiration of invertebrates, disruption of ichthyoplankton development, reduction of growth and survival of filter feeders, and decreased foraging efficiency of sight-feeders (Messieh et al. 1991; Barr 1993).

Introduction of pathogens

Pathogens are generally a concern to human health because of consumption of contaminated shellfish and finfish and exposure at beaches and swimming areas (USEPA 2005). Microorganisms entering aquatic habitats in sewage effluents do pose some level of biological risk since they have been shown to infect marine mammals (Oliveri 1982; Bossart et al. 1990; Islam and Tanaka 2004). The degree to which anthropogenically-derived microbes may affect fish, shellfish, and other aquatic taxa remains an important research topic; however, some recently published observations concerning groundfish populations near the Boston sewage outfall into Massachusetts Bay are suggesting that appropriate management practices may address at least part of this risk (Moore et al. 2005). See also the chapters on Coastal Development and Introduced/Nuisance Species and Aquaculture for more information on the introduction of pathogens.

Introduction of harmful algal blooms

Sewage treatment facilities releasing effluent with a high BOD that may enter estuarine and coastal habitats have been associated with harmful algal bloom events, which can deplete the oxygen in the water during bacterial degradation of algal tissue and result in hypoxic or anoxic "dead zones" and large-scale fish kills (Deegan and Buchsbaum 2005). There is evidence that nutrient overenrichment has led to increased incidence, extent, and persistence of nuisance and/or noxious or toxic species of phytoplankton; increased frequency, severity, spatial extent, and persistence of hypoxia; alterations in the dominant phytoplankton species and size compositions; and greatly increased turbidity of surface waters from plankton algae (O'Reilly 1994).

Algal blooms may also contain species of phytoplankton such as dinoflagellates that produce toxins. Toxic algal blooms, such as red tides, can decimate large numbers of fish, contaminate shellfish beds, and cause health problems in humans. Shellfish sequester toxins from the algae and become dangerous to consume. Toxic algal blooms could increase in the future because many coastal and estuarine areas are currently moderately to severely eutrophic (Goldburg and Triplett 1997). Heavily developed watersheds tend to have reduced stormwater storage capacity, and the high flow velocity and pulse of contaminants from freshwater systems can have long-term, cumulative impacts to estuarine and marine ecosystems. Some naturally occurring microorganisms, such as bacteria from the genus, *Vibrio*, or the dinoflagellate, *Pfiesteria*, can produce blooms that release toxins capable of harming fish and possibly human health under certain conditions (Buck et al. 1997; Shumway and Kraeuter 2000). Although the factors leading to the formation of blooms for these species will require additional research, nutrient enrichment of coastal waters is suspected to play a role (Buck et al. 1997). See also the chapter on Introduced/Nuisance Species and Aquaculture for more information on harmful algal blooms.

Impacts to benthic habitat

As discussed above, treated sewage effluent containing high concentrations of nutrients can lead to severely eutrophic conditions that can reduce or eliminate SAV beds (Goldsborough 1997). In addition, municipal sewage outfalls can release suspended sediments into the water column and the adjacent benthic habitat. Increased suspended particles within aquatic habitat can cause elevated turbidity levels, reduced light transmittance, which may lead to the reduction or loss of SAV, shellfish beds and other productive benthic habitats.

Changes in species composition

Treated sewage effluent can contain, at various concentrations, nutrients, toxic chemicals, and pathogens that can affect the health, survival, and reproduction of aquatic organisms. These effects may lead to alterations in the composition of species inhabiting coastal aquatic habitats and can result in community and trophic level changes (Kennish 1998). For example, highly eutrophic water bodies have been found to contain exaggerated phytoplankton and macroalgal populations that can lead to harmful algal blooms (Anderson et al. 2002). Sewage treatment facilities may initially attract fish around the point of discharge until hypoxia, toxin production, and algal bloom development render the aquatic area less productive (Islam and Tanaka 2004). Reduced light penetration in the water column from nutrient enrichment and sedimentation has been shown to contribute to the loss of eelgrass beds in coastal estuaries in southern Massachusetts, Long Island Sound, and the Chesapeake Bay (Goldsborough 1997; Deegan and Buchsbaum 2005).

Contaminant bioaccumulation and biomagnification

Sewage discharges can contain metals and other substances known to be toxic to marine organisms. Not surprisingly, the bays and estuaries of highly industrialized urban areas in northeastern US coastal areas, such as Boston Harbor, Portsmouth Harbor, NH/ME, Newark Bay,NJ, western Long Island Sound, and New York Harbor, have shown relatively high metal burdens in sampled sediments (Larsen 1992; Kennish 1998; USEPA 2004a). While industrial outfalls are responsible for metal contamination in some areas, sewage has been identified as one of the primary sources. For example, although lead contamination in coastal sediments can originate from a variety of sources, sewage is believed to be the primary source of silver contamination (Buchholtz ten Brink et al. 1996). Metals may move upward through trophic levels and accumulate in fish and some invertebrates (bioaccumulation) at levels which can eventually cause health problems in human consumers (Kennish 1998; NEFMC 1998). Other chemicals are known to bioaccumulate and biomagnify in the ecosystem, including pesticides (e.g., DDT) and PCB congeners (Kennish 1998). The National Coastal Condition Report (USEPA 2004a) reported that after metals, PCB congeners and DDT metabolites were responsible for most of the contaminant criteria exceedances in northeast coast samples. For example, sediment samples collected by NOAA's National Status and Trends (NS&T) Program found in some samples very high concentrations of chlorinated hydrocarbons such as PCBs, pesticides, and dioxins from the lower Passaic River, NJ, and Newark Bay in the Hudson-Raritan estuary (Long ER et al. 1995). Other locations in this estuary containing moderately to highly toxic samples in the NS&T Program included Arthur Kill, NY/NJ, and East River, NY.

Release of pharmaceuticals

Concerns have been emerging over the past few years regarding the continual exposure of aquatic organisms to the complex spectrum of active ingredients in pharmaceuticals and personal care products (PPCP), which can persist in treated effluent from sewage facilities. PPCPs comprise thousands of chemical substances, including prescription and over-the-counter therapeutic drugs, veterinary drugs, fragrances, lotions, and cosmetics (Daughton and Ternes 1999; USEPA 2007). The concentrations of PPCP in the aquatic environment are generally detected in the range of parts per thousand to parts per billion and may not pose an acute risk. However, aquatic organisms may be adversely affected because they can have continual and multigenerational exposures, exposures at high concentrations from untreated water, and they may have low dose effects (Daughton and Ternes 1999; USEPA 2007). Some of these PPCPs include steroid compounds, which may act as endocrine disruptors by mimicking the functions of sex hormones (refer to the subsection below for more information on endocrine disruptors). The effects of antibiotics and antimicrobial drugs on aquatic organisms are also of concern. Although population level effects on aquatic organisms from PPCPs are inconclusive at this time, the growing evidence on this topic suggests further investigation is warranted.

Endocrine disruptors

Another recent topic of concern involves a group of chemicals, called "endocrine disruptors," which interfere with the endocrine system of aquatic organisms. Growing concerns have mounted in response to the effects of endocrine-disrupting chemicals on humans, fish, and wildlife (Kavlock et al. 1996; Kavlock and Ankley 1996). These chemicals act as "environmental hormones" that may mimic the function of the sex hormones androgen and estrogen (Thurberg and Gould 2005). Adverse effects include reduced or altered reproductive functions, which could result in population-level impacts. Several studies have implicated endocrine-disrupting chemicals with the presence of elevated levels of vitellogellin in male fish, a yolk precursor protein that is normally only found in mature female fish (Thurberg and Gould 2005). Some of the chemicals shown to be estrogenic include PCB congeners, dieldrin, DDT, phthalates, and alkylphenols (Thurberg and Gould 2005), which have had or still have applications in agriculture and may be present in irrigation water and storm water runoff. Metals have also been implicated in disrupting endocrine secretions of marine organisms, potentially disrupting natural biotic processes (Brodeur et al. 1997).

In summary, the chemical implications of sewage treatment plant effluents vary as a function of the effort taken to remove organic and inorganic contaminants collected by the wastewater treatment plant. Further complicating matters, while secondary treatment is the minimal acceptable standard treatment process in the northeastern United States, inadequately treated or even raw wastewater containing human sewage and attendant debris routinely passes into the aquatic environment from municipal processing plant outfalls when the flow and/or storage demands exceed design specifications. Such releases are commonly experienced when older sewer systems are inundated, particularly in conjunction with storm events. Accordingly, the types of treatment processes implemented, how effectively the wastewater treatment infrastructure is operating, and the salinity of the receiving waters (to the extent that it influences contaminant chemistry) are critical variables when considering the chemical implications of releasing treated wastewater into the aquatic environment.

Maintenance activities associated with sewage discharge facilities

Maintenance activities associated with sewage treatment plants typically involve periodic application of chemicals to treat piping for colonization of biofouling organisms. Efforts to control fouling communities can produce larger field or even chronic disturbances that could adversely affect the aquatic environment. Under some circumstances, chemical treatments are not necessary and fouling communities may be removed mechanically using hot water under pressure. When this type of procedure is implemented, most of the direct impacts are physical. Although the use of pressurized, hot freshwater to mechanically remove fouling organisms may temporarily alter salinity and solute loads, some localized indirect thermodynamic effects that alter ambient chemistry could also occur in the dispersal plume until ambient temperature is restored. In addition, differences in the chemical composition of the source and receiving waters would be expected to have at least a minimal effect, particularly when chlorinated water is used to facilitate the removal of fouling organisms and when there is a significant difference in salinity between cleaning and receiving waters. Perhaps more typically, colonization of fouling communities is controlled through periodic use of antifouling paints, coatings, or other treatments. When conducted inappropriately, periodic applications of these substances can have chronic and potentially harmful effects in the aquatic environment.

Fortunately, application of biocides in aquatic systems is regulated under the CWA, which includes provisions to protect fishes and many invertebrate species to the extent practicable. Since local salinity ranges and diffusion rates at the outfall are important considerations in terms of eventual dispersion and relative toxicity of outfall maintenance materials, these and similar site-specific considerations often dictate which products may be used safely at a given project site. It is vital that only products designed and federally approved for use in and near aquatic habitats are deliberately allowed to enter US waterways under any circumstances.

In general, the most deleterious effects of sewage outfall maintenance probably revolve around fouling community control measures. That is because the underlying intent of such practices is to remove a large variety of plant, animal, and even bacterial populations from inhabiting the area surrounding the outfall. Biocide applications control undesirable organisms by chemical or biological means (Knight and Cooke 2002). Whether removed chemically or mechanically, the loss of these organisms at least initially may result in other forms of local ecological disturbance, such as reduced productivity and diminished prey and cover (Meffe and Carroll 1997). While outfall maintenance events individually result in an acute chemical impact to the environment and biota, it is important also to consider the cumulative effects of repeated applications over a project's maintenance cycle. Especially when undertaken regularly, the maintenance of outfall structures can create a chronic cycle of disturbance on resident biota, particularly sessile organisms.

Individual biocides and other contaminants released during outfall maintenance operations may have direct effects on local aquatic biota or they may act in an additive, synergistic, or antagonistic manner in concert with ambient physical and chemical habitat conditions. Such exposure to organic and inorganic pollutants may result in a spectrum of lethal and sublethal effects that may be discerned at every level of biological organization (Thurberg and Gould 2005). Wide distribution of contaminants, such as biocides and related outfall maintenance substances, can be facilitated through bioaccumulation in motile aquatic organisms that are capable of dispersing between riverine, estuarine, and marine habitats (Mearns et al. 1991). The pollutant-induced effects these substances engender are not limited to biochemical or physiological responses, as they may also disrupt a variety of complex behaviors which may be essential for maintaining fitness and survival (Atchison et al. 1987; Blaxter and Hallers-Tjabbes 1992; Kasumyan 2001; Scott and Sloman 2004).

In addition to measures to control fouling organisms in wastewater treatment facilities, maintenance activities also involve repairs and enhancements of structures associated with the facilities' infrastructure. Because they typically are undertaken on a relatively small scale, physical repairs of existing infrastructure usually produce impacts of lesser intensity and on a more limited spatial scale than those created during initial installation. In contrast, application of antifouling coatings or related treatments not only discourages settlement by aquatic organisms on the treated surface, but also releases biocide into the aquatic environment (Richardson 1997; Terlizzi et al. 2001). Depending on the individual case, such releases can range from very limited to extensive plumes, as measured by the volume of material emitted, and the distance broadcast away from the point source the substance may be detected in the water column.

Collectively, such releases degrade local water quality. Fortunately, chemical effects of sewage outfall maintenance in lotic coastal systems generally would be expected to dissipate relatively quickly because of dispersion by river flow or tidal action. For health and aesthetic reasons, municipal sewage outfalls should not be sited in quiescent waters. In addition, government-established protocols for biological control agents approved for applications in subaqueous discharges generally are applied in isolation within a capped pipe and are subsequently released after sufficient time has passed for the biocide properties to have abated, or more rarely after the bulk of the treating solution is siphoned off and dealt with offsite. Typically, such biocide solutions are designed to decompose into relatively benign constituent forms within hours and, when used properly, are thought not to pose a significant risk to nontarget organisms (Diderich 2002).

As is the case for initial outfall installation impacts, a variety of chemical and biological factors determine the extent to which the polluting substance affects the water column, sediments, and biota and the distance it migrates from the point source. Among them, salinity and carbonate alkalinity (i.e., carbonic acid and bicarbonate ion content) are especially important because of their respective roles in mediating chemical reactions in solution and in conferring the buffering capacity provided by marine and estuarine waters. Carbonate alkalinity, or water hardness, is an especially important property in riverine systems because the ambient carbonate concentrations regulate acid-base chemistry and other water quality parameters, which are thought to be important factors in the recovery of depleted salmonid populations in Maine (Johnson and Kahl 2005). While salmonids are particularly sensitive to degraded water quality, poor water quality is known to affect a wide variety of aquatic organisms (Tessier et al. 1984; Scott and Sloman 2004; Moore et al. 2005; Thurberg and Gould 2005).

Construction impacts associated with sewage discharge facilities

The construction of municipal wastewater outfalls can have chemical effects that result from a number of activities, including releasing suspended sediments and associated pore-water in the construction zone; releasing drill mud or cuttings from a directional drilling operation; discharging substances from mechanized equipment (e.g., incidental discharges of hydrocarbons or hydraulic fluid); and introducing leachate from fresh and curing concrete, antifouling paints, and other construction materials. Contaminants initially reside in aquatic systems in either a dissolved phase in the water column or in a particulate phase when they have adsorbed onto sediments or other solids. Pollutants present in biologically-available forms subsequently become assimilated by aquatic biota and become biomagnified as they are taken up in successive trophic strata (Levinton 1982; Sigel and Sigel 2001).

While plume and sedimentation effects incidental to outfall construction do not always result in a readily observable ecological response, they commonly produce a range of direct and indirect effects to living aquatic resources and their habitats. Not all of the ecological implications of sediment resuspension and transport result in adverse effects to aquatic organisms (Blaber and Blaber 1980). These effects vary a great deal depending on which life history stages are affected (Wilber and Clarke 2001). As a general rule, however, the severity of adverse chemical effects tends to be greatest for early life stages and for adults of some highly sensitive species (Newcombe and Jensen 1996). In particular, predictive models of dose-response relationships corroborate that the eggs and larvae of nonsalmonid estuarine fishes exhibit some of the most sensitive responses to suspended sediment exposures of all the taxa and life history stages for which data are available (Wilber and Clarke 2001). Mitigative measures that limit the nature and extent of chemical impacts arising from outfall installation typically can and should be undertaken to avoid and minimize adverse construction effects.

From the standpoint of water quality, most chemical effects associated with outfall construction should be relatively acute and transitory. Adverse water quality impacts arising from outfall installation generally arise as a consequence of: (1) substances that have adsorbed onto resuspended particles; (2) pollutants that have dissolved or leached into the water column; or (3) contaminants that have been released directly by construction equipment. These pollutants may include substances that lead to nutrient enrichment; they may be chemically reduced; they may exhibit acidic or caustic properties; they may contain organometallic complexes or a variety of other natural or synthetic compounds; they may be hydrophobic or hydrophilic; or they otherwise may exhibit a diverse spectrum of chemical properties that affect their relative toxicity and dispersal in the water column.

While various physical, chemical, and biological factors come into play, the area into which such water quality impacts extend is largely dependent upon the length of time particles and solutes are held in the water column and the distance they are transported from the construction site. Grain size and ambient sediment structure characteristics have an important bearing on dispersal. As benthic material is disturbed during outfall installation and site preparation, resuspended particulate matter would settle predominantly in the immediate project vicinity. Remaining waterborne fractions subsequently would be transported over a distance and direction that are related to the grain size of disturbed sediments, the velocity of local water currents, and local wave action (Neumann and Pierson 1966). Contaminants mobilized in and subsequently deposited by the dispersal plume generated by construction activities are subject to complex biogeochemical processes that ultimately dictate their fate and ecological effects. For example, hydrogen sulfide released with pore-water from disturbed sediments depletes dissolved oxygen and results in locally hypoxic or anoxic conditions in the water column until the area engulfed within the dispersal plume becomes reoxygenated.

While important, it is essential to recognize that local sediment characteristics alone do not determine contaminant introduction or resuspension during outfall installation. The type of construction equipment used to build an outfall structure also has an important influence on the dispersion of disturbed bottom material. For traditional clamshell dredging, Tavolaro (1984) estimates a 2% loss of material through sediment resuspension at the dredge site. It is reasonable to conclude that similar losses would accrue when clamshells are used to install outfall pipes for sewage treatment facilities. In the same way, dredging methods that purposely fluidize sediments to facilitate their removal (e.g., hydraulic dredges, water jets) could result in even greater dispersion of resuspended sediment, especially when local waters are not quiescent or in situations where unfiltered return flow to the waterway is permitted. Since fine depositional sediments tend to have greater contaminant loads than do coarser sediments typical of higher energy areas, the chemical consequences of resuspending fine sediments during outfall installation are potentially greater since they are more likely to be associated with pollutants.

Likewise, water quality implications of outfall construction are not limited to sediment resuspension or releasing pore-water that contains hydrogen sulfide. Secondary vectors of chemical contamination during outfall installation include substances introduced into aquatic habitats by construction equipment and materials. Mechanized construction equipment may inadvertently or incidentally release a broad spectrum of chemicals, fuels, and lubricants into the waterway. Similarly, until the building material has completely cured or has leached out soluble contaminant fractions, subaqueous applications of wet concrete or grout, treated timber products, paints, and other construction materials would all potentially introduce pollutants into the surrounding water.

The chemical implications of constructing municipal outfalls to local substrates ultimately depend on whether (and to what extent) contaminants are released, become associated with, and accumulate in, sediments and surrounding pore-water. While sediment particles naturally exhibit cycles of exchange between the water column and bottom substrate materials (Turner and Millward 2002), dredging or outfall installation can be expected to disturb much deeper sediment horizons in a short period of time than would be expected from storms or in all but the most highly erosion prone coastal areas. As construction equipment disrupts sediment horizons at the project site, some fraction of the benthic substrate becomes resuspended into the water column (Tavolaro 1984).

Outfall construction for sewage treatment facilities can create measurable adverse impacts within the disturbed footprint, including the disruption of ambient sediment stratigraphy, cohesiveness, and geochemistry. These effects have geochemical consequences that may be particularly significant when construction activities are located in depositional or nutrient-enriched areas and where local sediments tend to be fine-grained and contain at least moderate levels of pollution. Regardless of the nature and concentration of substances adsorbed onto the sediment or sequestered in the pore-water, salinity may significantly affect local aqueous conditions, sedimentary geochemistry, and resulting ecological effects.

While it is critical to consider the impacts of outfall construction on physical habitat features, implications for resident and transitory biota also should be taken into account. Excavation and relocation of sediments, which may be performed incidental to outfall installation, would produce a sediment plume and create sedimentation effects that could result in detrimental effects on aquatic resources present in the affected area (Newcombe and Jensen 1996; Wilber and Clarke 2001; Berry et al. 2003; Wilber et al. 2005). Direct and indirect impacts related to the removal of benthic material can elicit a variety of responses from aquatic biota (Wilber and Clarke 2001) which have been addressed elsewhere in this report.

While many potential construction impacts clearly are physical in nature, the chemical effects are complex and may have important implications for biota present in the affected area. In addition to the physicochemical considerations already discussed above, the life history and ecological strategies characteristic of different species also are important considerations in assessing the potential chemical impacts of outfall installation. For instance, while highly motile adult and fish in juvenile life stages of most species could flee when construction is ongoing, those in egg and larval stages and nonmotile benthic organisms could not escape contaminant exposure. While some species like the sessile life stages of eastern oyster (*Crassostrea virginica*) have adapted to withstand some acute habitat disturbances (Galtsoff 1964; Levinton 1982), most benthic and slow-moving species would not be able to escape contaminant exposure and instead would exhibit adaptive physiological and biochemical responses to counter any pollutants present.

Contaminants released during outfall installation activities may have direct effects on local aquatic biota or they may act in an additive, synergistic, or antagonistic manner in concert with ambient physical and chemical habitat conditions. Such exposure to organic and inorganic pollutants may result in a spectrum of lethal and sublethal effects that can be discerned at the organismal, tissue, cellular, and subcellular levels of biological organization (Thurberg and Gould 2005). Wide distribution of contaminants can be facilitated through bioaccumulation in motile aquatic organisms that are capable of dispersing between riverine, estuarine, and marine habitats (Mearns et al. 1991).

Importantly, pollutant-induced effects are not limited to biochemical or physiological responses. Environmental pollutants such as metals, pesticides, and other organic compounds also have been shown to disrupt a variety of complex fish behaviors, some of which may be essential for maintaining fitness and survival (Atchison et al. 1987; Blaxter and Hallers-Tjabbes 1992; Kasumyan 2001; Scott and Sloman 2004). In particular, Kasumyan (2001) provided an excellent review of how chemical pollutants interfere with normal fish foraging behavior and chemoreception physiology, while Scott and Sloman (2004) have focused on the ways metals and organic pollutants have been shown to induce behavioral and physiological effects on fresh water and marine fishes.

Industrial Discharge Facilities

Introduction

Industrial wastewater facilities face many of the same engineering and environmental challenges as municipal sewage treatment plants. Industrial discharge facilities produce a wide variety of trace elements and organic and inorganic compounds. In the industrialized portions of the northeastern United States, such facilities include a variety of chemical plants, refineries, paper mills, defense factories, energy generating facilities, electroplating firms, mining operations, and many other high intensity industrial uses that generate large volumes of wastewater. In many situations, the sanitary and industrial process streams are intermingled and processed at the industrial facility's own treatment plant, requiring that the eventual effluent is treated to address water quality concerns from a fairly broad spectrum of contaminants. While the procedures involved are similar to those implemented at municipal treatment facilities, the specific levels and methods of wastewater treatment at industrial treatment plants vary considerably. While a detailed description of industrial wastewater engineering is well beyond the scope of this report, readers interested in specific technical information may consult portions of Tchobanoglous et al. (2002) or Perry (1997) for more information.

Like sewage plant outfalls, industrial discharge structures are point sources for a variety of environmental contaminants, particularly metals and other trace elements; nutrients; and persistent organic compounds such as pesticides and organochlorines. These substances tend to adhere to solid particles within the waste stream, become adsorbed onto finer sediment fractions once dispersed into coastal waters, and subsequently accumulate in depositional areas. Together with microbial action, local salinity and other properties of the riverine, estuarine, or marine receiving waters may alter the chemistry of these contaminant-particle complexes in ways that render them more toxic than their parent compounds. Upon entering the food web, such contaminants tend to accumulate in benthic organisms at higher concentrations than in surrounding waters (Stein et al. 1995) and may result in various physiological, biochemical, or behavioral effects (Scott and Sloman 2004; Thurberg and Gould 2005).

Release of metals

Industrial discharge structures can release large volumes of effluent containing a variety of potentially harmful substances into the aquatic environment. Metals and other trace elements are common byproducts of industrial processes and as a consequence are anticipated to be components of typical industrial waste streams that may enter the aquatic environment (Kennish 1998). Metals may be grouped into transitional metals and metalloids. Transitional metals, such as copper, cobalt, iron, and manganese, are essential for metabolic function of organisms at low concentrations but may be toxic at high concentrations. Metalloids, such as arsenic, cadmium, lead, mercury, and tin, are generally not required for metabolic function and may be toxic even at low concentrations (Kennish 1998). Metals are known to produce skeletal deformities and various developmental abnormalities in marine fish (Bodammer 1981; Klein-MacPhee et al. 1984; Lang and Dethlefsen 1987). The early life history stages of fish can be quite susceptible to the toxic impacts associated with metals (Gould et al. 1994).

Release of organic compounds

A variety of synthetic organic compounds are released by industrial facilities, find their way into aquatic environments and can be taken up by resident biota. These compounds are some of the most persistent, ubiquitous, and toxic pollutants known to occur in marine ecosystems (Kennish 1998). Organochlorines, such as DDT, chlordane, and PCBs, are some of the most highly toxic, persistent, and well documented and studied synthetic organic compounds. Others include dioxins and dibenzofurans that are associated with pulp and paper mills and wood treatment plants and have been shown to be carcinogenic and capable of interfering with the development of early development stages of organisms (Kennish 1998). Longwell et al. (1992) determined that dozens of different organic contaminants were present in ripe winter flounder eggs. Such accumulation can reduce egg quality and disrupt ontogenic development in ways that significantly depress survival of young (Islam and Tanaka 2004). Organic contaminants, such as PCBs, have been shown to induce external lesions (Stork 1983) and fin erosion (Sherwood 1982) and reduce reproductive success (Nelson et al. 1991) in marine fishes. In addition, suspicion is mounting that exposure to even very low levels of such persistent xenobiotic (i.e., foreign) compounds may disrupt normal endocrine function and lead to reproductive dysfunction such as reduced fertility, hatch rate, and offspring viability in a variety of vertebrates.

Release of petroleum products

Oil, characterized as petroleum and any derivatives, consists of thousands of chemical compounds and can be a major stressor on inshore fish habitats (Kennish 1998). Industrial wastewater, as well as combined wastewater from municipal and storm water drains, contributes to the release of oil into coastal waters. Petroleum hydrocarbons can adsorb readily to particulate matter in the water column and accumulate in bottom sediments, where they may be taken up by benthic organisms (Kennish 1998). Petroleum products consist of thousands of chemical compounds that can be toxic to marine life including PAHs and water-soluble compounds, such as benzene, toluene, and xylene, which can be particularly damaging to marine biota because of their extreme toxicity, rapid uptake, and persistence in the environment (Kennish 1998). PAHs can be toxic to meroplankton, ichthyoplankton, and other pelagic life stages exposed to them in the water column (Kennish 1998). Short-term impacts include interference with the reproduction, development, growth, and behavior (e.g., spawning, feeding) of fishes, especially early life-history stages (Gould et al. 1994). Oil has been demonstrated to disrupt the growth of vegetation in estuarine habitats (Lin and Mendelssohn 1996). Although oil is toxic to all marine organisms at high concentrations, certain species are more sensitive than others. In general, the early life stages (eggs and larvae) are most sensitive, juveniles are less sensitive, and adults least so (Rice et al. 2000). Refer to the chapters on Coastal Development, Energy-related Activities, and Marine Transportation for additional information on impacts associated with petroleum products and PAH.

Alteration of water alkalinity

A major point of departure when comparing municipal sanitary treatment outfall and industrial plant effluents concerns the ability of some industrial discharges to affect carbonate alkalinity, or buffering capacity, of receiving waters. Both riverine and estuarine strata are particularly susceptible to point source acidification because their low buffering capacity can be quickly overwhelmed by acid discharges; however, even marine habitats can be significantly and adversely affected when continual influx of acidified liquid wastewater outstrips the natural buffering capability of seawater. In riverine systems, it has been postulated that locally reduced pH may be linked to impaired Atlantic salmon recovery (Johnson and Kahl 2005) and osmoregulatory problems (NRC 2004). Oulasvirta (1990) reported periodic massive mortalities of Atlantic herring eggs from effluent containing sulfuric acid and various other metals released at a titanium-dioxide plant in the Gulf of Bothnia, Finland. Low pH in estuarine waters may lead to cellular changes in muscle tissues, which could reduce swimming ability in herring (Bahgat et al. 1989). A variety of industrial operations, ranging from mining and metal production to certain industrial manufacturing activities, is known to release acid effluents that may have adverse effects on fish, shellfish, and their habitat. Collectively, such detrimental impacts can hinder the survival and sustainability of fishery resources and their prey. Point source pollution from industrial sources is currently regulated by the states or the US EPA through the NPDES permit program, which generally does not allow discharges of low pH water into estuaries and coastal waters of the United States.

Release of nutrients and other organic wastes

Industrial facilities that process animal or plant by-products can release effluent with high BOD which may have deleterious affects to receiving waters. Wood processing facilities, paper and pulp mills, and animal tissue rendering plants can release nutrients, reduced sulfur and organic compounds, and other contaminants through wastewater outfall pipes. For example, wood processing plants and pulp mills release effluents with tannins and lignin products containing high organic loads and BOD into aquatic habitats (USFWS and NMFS 1999). The release of these contaminants in mill effluent can reduce dissolved oxygen in the receiving waters. In addition, paper and pulp mills can release a number of toxic chemicals used in the process of bleaching pulp for printing and paper products. The bleaching process may use chlorine, sulfur derivatives, dioxins, furans, resin acids, and other chemicals that are known to be toxic to aquatic organisms (Mercer et al. 1997). These chemicals have been implicated in various abnormalities in fish, including skin and organ tissue lesions, fin necrosis, gill hyperplasia, elevated detoxifying enzymes, impaired liver functions, skeletal deformities, increased incidence of parasites, disruption of the immune system, presence of tumors, and impaired growth and reproduction (Barker et al. 1994; Mercer et al. 1997). Because of concern about the release of dioxins and other contaminants, considerable improvements in the bleaching process have reduced or eliminated the use of elemental chlorine. According to the US EPA, all pulp and nearly all paper mills in the United States have chemical recovery systems in place and primary and secondary wastewater treatment systems installed to remove particulates and BOD (USEPA 2002). Approximately 96% of all bleached pulp production uses chlorine-free bleaching technologies (USEPA 2002).

Construction impacts of industrial discharge facilities

The chemical impacts associated with constructing an industrial discharge are similar to those described for sewage treatment outfalls. Generally, such discharges predominantly entail suspending sediments and releasing pore-water in the construction zone, releasing drill mud or cuttings from horizontal directional drilling equipment, incidental discharges of fuels, lubricants and other substances from mechanized construction equipment, and leachates from construction materials. Since the substances encountered and circumstances of exposure would be the same regardless of the type of outfall being installed, the Construction Impacts Associated with Sewage Discharge Facilities subsection of this chapter should be reviewed for details regarding the impacts to the water column, sediment, and aquatic biota from the construction of industrial discharge facilities.

Maintenance impacts of industrial discharge facilities

The chemical impacts of maintaining industrial discharge facilities are similar to those described for sewage treatment facilities. Generally, the impacts of performing structural repairs are expected to be similar to those experienced during initial outfall installation, but on a lesser scope and magnitude. Impacts associated with the removal and treatment of fouling communities would be similar to those described for the maintenance activities of sewage treatment facilities. The reader should review the previous subsection on Maintenance Activities Associated with Sewage Discharge Facilities for details on the implications of outfall maintenance on the water column, sediment, and aquatic biota.

Combined Sewer Overflow (CSO)

The discussion of point source discharges would be incomplete without mention of CSOs, which are ubiquitous in urban and even suburban areas in New England and the Mid-Atlantic region. For a variety of reasons, many of these municipalities operate wastewater collection systems composed of "separate" and "combined" sewers. "Separate" sewers tend to be newer or replacement installations that have distinct piping components for stormwater and sanitary sewers.

Under storm or other high runoff conditions, the separate sewer system allows excess volumes of storm water to bypass sewage treatment facilities and discharge directly into the receiving water body constraining all sanitary waste to processing at the wastewater treatment plant. This prevents the excess volume of watershed runoff from overwhelming the operating capacity of the treatment facilities. Older systems tend to be "combined" sewer systems that commingle watershed runoff and sanitary waste streams.

Typical CSOs do not discharge effluent under dry conditions but may permit unprocessed sewage under high runoff events to enter the receiving waters completely or partially untreated. This occurs when large volumes of storm water and sewage overwhelm the treatment plant and untreated sewage is discharged prematurely. Some CSO discharges violate state and/or federal water quality standards, and each municipality must develop a plan to control and eliminate these CSOs. There is no precise estimate on the number of CSOs that exist or on how much untreated sewage is discharged from them each year. However, 828 separate NPDES permits were issued by the US EPA in 2004. There were a total 9,348 authorized discharges from CSOs nationally in 2004, with approximately one half located in the northeastern United States and the remaining half in the Great Lakes region (USEPA 2002; USEPA 2004b).

The chemical implications of CSOs are that they are potential sources of very large amounts of untreated nutrients and contaminating chemicals that degrade both the aesthetic and ecological conditions of affected habitats. In addition to the adverse effects mentioned for the other outfall types, CSOs can be important point sources for pesticides, herbicides, fertilizers, and other substances commonly applied to terrestrial habitats, ranging from rural farmland and suburban yards or golf courses to highly urbanized centers. In addition, they are sources of terrestrial particulates and may be a secondary source of atmospherically-deposited pollutants that have settled anywhere in the local watershed. While impacts associated with nonpoint sources are discussed elsewhere in this report, the sanitary sewer component of CSO effluents can be construed as an extension of the preceding discussions for municipal and industrial outfalls. The net effect of permitting untreated domestic wastewater to enter the receiving waterway is to diminish the effectiveness of wastewater treatment elsewhere. In so doing, CSOs contribute to increased pollution levels and related natural resource impairments. It is not possible to measure the resulting habitat damage and accompanying aquatic resource degradation in isolation from nonpoint pollution. However, it is important that resource managers consider that CSO discharges can and will occur and account for the added pollutant loads they generate when setting permissible local effluent limits or establishing priorities for replacing outmoded urban infrastructure.

Construction and maintenance impacts of CSOs

The chemical impacts associated with construction and maintenance activities in CSOs are similar to those described for sewage treatment and industrial discharge facilities. Generally, discharges associated with construction activities may include releasing contaminants associated with suspended sediments, releasing pore-water and drill mud or cuttings from directional drilling, discharges of fuels, lubricants, and other substances from construction equipment. Maintenance activities may include the removal and treatment of fouling communities and releases of contaminants similar to those described above. The reader should refer to the Construction Impacts Associated with Sewage Discharge Facilities and the Maintenance Activities Associated with Sewage Discharge Facilities subsections of this chapter for additional information on this topic.

Conservation measures and best management practices for sewage and industrial discharge facilities and CSOs (adapted from Hanson et al. 2003)

1. Locate discharge points in coastal waters well away from shellfish beds, submerged aquatic vegetation, reefs, fish spawning grounds, and similar fragile and productive habitats.
2. Determine benthic productivity by sampling prior to any construction activity related to installation of new or modified facilities. Implement all appropriate best management practices to maintain habitat quality during construction including any seasonal restrictions, use of cofferdams, working in the dry at low tide, etc., as is necessary and practicable.
3. Use seasonal restrictions during construction and maintenance operations to avoid impacts to habitat during species' critical life history stages (e.g., spawning and egg development periods), when appropriate. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
4. Develop appropriate modeling studies for plume effects and other parameters of concern in cooperation with the involved resource agencies before finalizing outfall design. Any appropriate recommendations that involve agencies and developed as a consequence of the study results should be incorporated in the construction plans and operation plan for these facilities as enforceable permit conditions.
5. Institute all appropriate source control measures and/or elevate the treatment level to reduce the polluting substances in all effluents to the extent practicable. Ensure that discharge facilities obtain and adhere to NPDES program permits, as appropriate.
6. Ensure that maximum permissible discharges are appropriate for the given project setting and specify any and all operation procedures, performance standards, or best management practices that must be observed to address all reasonably foreseeable contingencies over the life of the project. Consider implementing an adaptive management plan that includes representatives from appropriate agencies to participate in future consultations for administering the management plan. Management plans should include monitoring protocols designed to measure discharge and potential impacts to sensitive resources and habitats.
7. Use best available technologies to treat discharges to the maximal effective and practicable extent, including measures that reduce discharges of biocides and other toxic substances.
8. Take precautions to mitigate the ecological damage arising from outfall maintenance activities. Facility maintenance plans should include measures such as: (a) ensuring biocides selected for a particular application are specifically designed for their intended use; (b) applying no more than the minimal effective dose, and; (c) closely following instructions for use in aquatic applications and ultimate disposal.
9. Use land treatment and upland disposal or storage for any sludge or other remaining wastes after wastewater processing is concluded. Use of vegetated wetlands as biofilters and pollutant assimilators for large-scale discharges should be limited only to circumstances where other less damaging alternatives are not available and the overall environmental suitability of such an action has been demonstrated.
10. Avoid locating pipelines and treatment facilities in wetlands and streams. Discharges should not be sited near eroding waterfronts or where receiving waters cannot reasonably assimilate the amount of anticipated discharge.
11. Ensure that the design capacity for all facilities will address present and reasonably foreseeable needs and that the best available technologies are implemented.
12. Encourage communities to reduce the volume of pollutants entering CSOs and reduce the number of CSO overflows during storm water runoff producing events. The US EPA provides recommended best management practices for communities (USEPA 1999), including: (a) reduce

and manage solid wastes streams; (b) encourage waste reduction and recycling; (c) reduce commercial and industrial pollution; (d) implement regular program of street cleaning; (e) maintain catch basins; (f) conserve water; (g) reduce unnecessary fertilizer and pesticide applications and; (h) control sediment and erosion.

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