6.13 Wastewater Pollution Impacts on Estuarine and Marine Environments

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Abstract

Wastewater pollution is a ubiquitous threat to the health of marine and estuarine ecosystems, yet it has been severely underestimated in the past. In light of the global sanitation crisis, growing water quality concerns, and rapid aging of wastewater infrastructure worldwide, wastewater inputs have come under greater scrutiny because they are now known to introduce problematic amounts of nutrients, pathogens, and novel contaminants into waterways. Although there have been a few comprehensive investigations of wastewater outfall impacts on nearshore and coastal waters in the past, scientists and environmental managers have increasingly begun to expose the repercussions of wastewater pollution on marine and coastal environments in recent years. This cross-ecosystem synthesis details the extent of domestic wastewater impacts, from individual organisms to ecosystem functions to global trends, and demonstrates a need for a paradigm shift towards sustainable wastewater management.

Key Points

- Clouded by misconceptions and minimal information, wastewater pollution has historically gone unchecked and only recently has gained attention from the broader scientific community and the public. Although some point sources of wastewater effluent are subject to strict regulations and have well-documented effects from routine monitoring programs, those cases are in the minority. Wastewater monitoring is lacking in most locations and oftentimes constrained by its high variability. Further, some treatment systems are deliberately designed to release raw sewage when overburdened by weather events, and in others, wastewater is collected only to be directly disposed of thereafter. Adding to the problem, nonpoint source inputs undergo even less scrutiny, if any at all, due to the complexity of the issue and difficulty in creating regulations.
- Wastewater effluent introduces novel contaminants which detrimentally affect ecological health, from the individual organism to the whole ecosystem level.
- This chapter focuses specifically on domestic wastewater; however, this should not detract from the gravity of impacts from other wastewater sources, such as industry and agriculture.
- Due to its global prevalence and severity, wastewater pollution has significant impacts on coastal and marine systems and affects coral reefs, seagrasses, mangroves, salt marshes, bivalve reefs, finfish, and marine mammals.
- Research gaps exist for determining the extent of wastewater impacts, especially since wastewater pollutants can act synergistically with each other and with external stressors (e.g., climate change).
- The emerging solution space addresses wastewater pollution via traditional and focused approaches (e.g., administrative controls, policy, governance, and ecosystem-based management strategies) to promote coastal protection, conservation and resilience, in addition to integrative techniques (e.g., nature-based solutions and advanced treatment technologies) to also tackle associated environmental goals (e.g., resource recovery and climate adaptation).

6.13.1 Introduction

Wastewater pollution can be found in coastal and marine environments around the world, regardless of an area's income level, the origin (point versus nonpoint sources), or the degree of treatment or sanitation infrastructure in place. Because wastewater pollution is relatively invisible and has impacts similar to those of agricultural runoff, its contribution to poor water quality and declining coastal habitat quality has been largely overlooked and misunderstood until recently (Tuholske *et al.*, 2021; Wear *et al.*, 2021).

Historically, the impacts of wastewater discharge on nearshore and marine environments have been studied to varying degrees, with researchers primarily focusing on impacts to habitats in proximity to large-scale wastewater treatment facilities' outfalls, as well as determining the environmental consequences of septic and cesspool failures. Although some coastal wastewater outfalls have well-documented effects from effective regulations and routine monitoring programs (e.g., Southern California Coastal Water Research Project), a recent review has found that only about half of 107 coral reef countries and territories have wastewater discharge standards, and many of those standards were developed without the specific intent of protecting ecological health (Wenger *et al.*, 2023). Marine water quality guidelines, which better address ecosystem health, exist in even fewer countries (Wenger *et al.*, 2023). Researchers have begun to conduct reviews and analyses to derive sediment and water quality standards that would protect coral health (Nalley *et al.*, 2023; Tuttle and Donahue, 2022), but country-wide standards have yet to reflect their findings. Considering this general gap in data and research, wastewater pollution has not received the same level of attention as other threats. Further, not only do basic and applied research issues and ecosystem-based management strategies pose a barrier, but societal and legal matters have also complicated and often hindered effective resolution of impacts. For instance, there have been legal challenges and pushback made against state and federal regulators by polluters – often in court. Other limiting factors include societal taboo, siloed sectoral efforts (i.e., lacking collaboration between research community and decision makers), lack of regulation, legal opposition to regulation, and few resources to enforce regulatory compliance; this leaves those working to mitigate wastewater pollution with limited information and only a partial understanding of wastewater composition. This latter limitation is particularly important given the long list of common pollutants in wastewater and their wide-ranging effects (Wear and Vega Thurber, 2015). In this chapter, we review the characteristics and components of wastewater pollution, global trends in its distribution, the growing presence of contaminants of emerging concern, and specific impacts of wastewater on a variety of estuarine and marine habitats and species.

First, it is important to establish what is meant by wastewater pollution. Here, we focus on domestic wastewater from both point and nonpoint sources. For example, point sources include sewage plants which process wastewater discharged from toilets, sinks, and showers from residential and commercial buildings and release it as effluent via outfalls, whereas nonpoint sources include discharges or leaks from on-site systems (e.g., pit latrines and cesspools) that gradually enter water bodies from the watershed. Point sources already prove difficult to survey and regulate, but nonpoint sources introduce even more complexity given their diffuse origin and spread. Domestic wastewater is not effluent from industrial or manufacturing activities of any kind, but it is just as significant a source of pollution for coastal and marine environments. Colloquially, the term “sewage” is often used in place of “wastewater”; however, the term “sewage” has a specific meaning in the sanitation industry and technically refers to water that flows through sewer pipes (often to a centralized treatment facility). In contrast, wastewater is a more expansive term that includes effluent from treatment plants, combined sewer overflows, septic systems, cesspools, pit latrines, and open defecation. Essentially, it includes all activities involving human excreta, cooking, cleaning, and personal hygiene.

A primary concern about wastewater pollution is that it often creates eutrophication problems for estuarine and marine habitats. Although agricultural fertilizer is frequently cited as the main culprit for eutrophication, wastewater is a significant contributor of excess nutrients in the environment (Tuholske *et al.*, 2021). Not to mention that wastewater contains a wide range of additional components that can be harmful to both habitats and species. The most common components of wastewater (**Fig. 1**) include inorganic nutrients, pathogens, endocrine disruptors, suspended solids, sediments, heavy metals, microplastics, freshwater, and chemical toxins such as per- and polyfluoroalkyl substances (PFAS) and polychlorinated biphenyls (PCBs) (Pantsar-Kallio *et al.*, 1999; Wear and Vega Thurber, 2015). Each of these components can have a variety of impacts on marine ecosystems and species, as well as synergistic effects when found in combination (Wear and Vega Thurber, 2015).

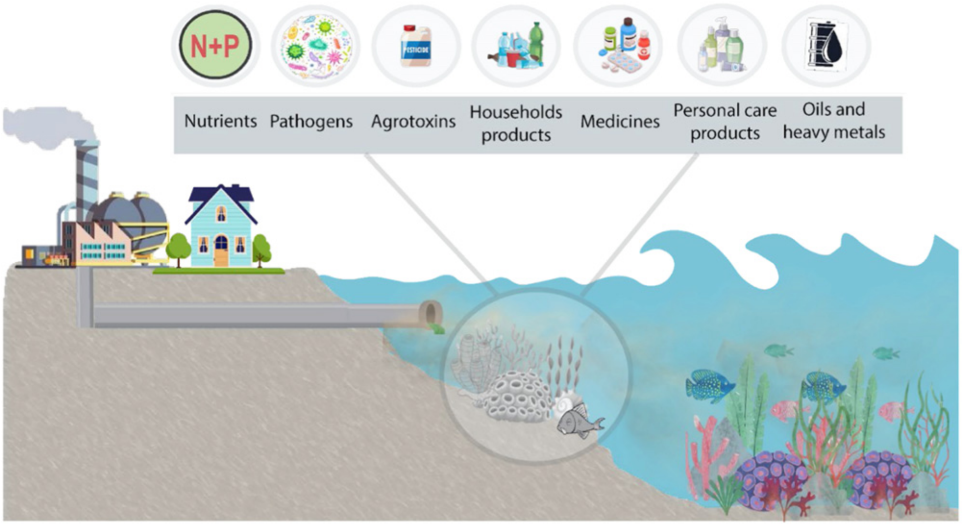

Fig. 1 Diagram depicting wastewater discharge to ocean waters and associated compounds that can negatively affect ocean health. Clockwise from the top, wastewater constituents can contain pesticides, plastics and microplastics, pharmaceuticals, microorganisms, petrochemicals, personal care products and household cleaning products, and nutrients. These constituents can have acute effects on nearshore marine life but can still impact distal ecological systems.

6.13.2 Spatial Extent, Patterns, and Composition of Wastewater Pollution In Coastal Ecosystems Worldwide

From small islands to the most populated coastlines on the planet, human wastewater is a global problem for marine ecosystems (Tuholske *et al.*, 2021). Densely populated areas along coastlines and within large watersheds in low- and middle-income countries (LMICs) tend to have the worst wastewater impacts on coastal ecosystems, but nearly all coastal areas experience some level of wastewater pollution (Tuholske *et al.*, 2021; Wear and Vega Thurber, 2015). Despite progress over the past two decades, two billion people, the vast majority of whom live in sub-Saharan Africa, still lack access to basic sanitation today (UNICEF and WHO, 2020); as such, much of the wastewater from LMICs flows into coastal ecosystems untreated (Tuholske *et al.*, 2021), with serious repercussions to ecosystem health. Nevertheless, depending on the treatment facilities, location of populations, and the type of pollutant, high-income countries can have equal or greater wastewater impacts per capita on coastal ecosystems (Tuholske *et al.*, 2021). Further, wastewater flows in small but densely-populated coastal watersheds in wealthy areas (e.g., the Hawaiian Islands) (Wada *et al.*, 2021), can significantly damage local coastal ecosystems and may not be resolved in coarse-grained global models.

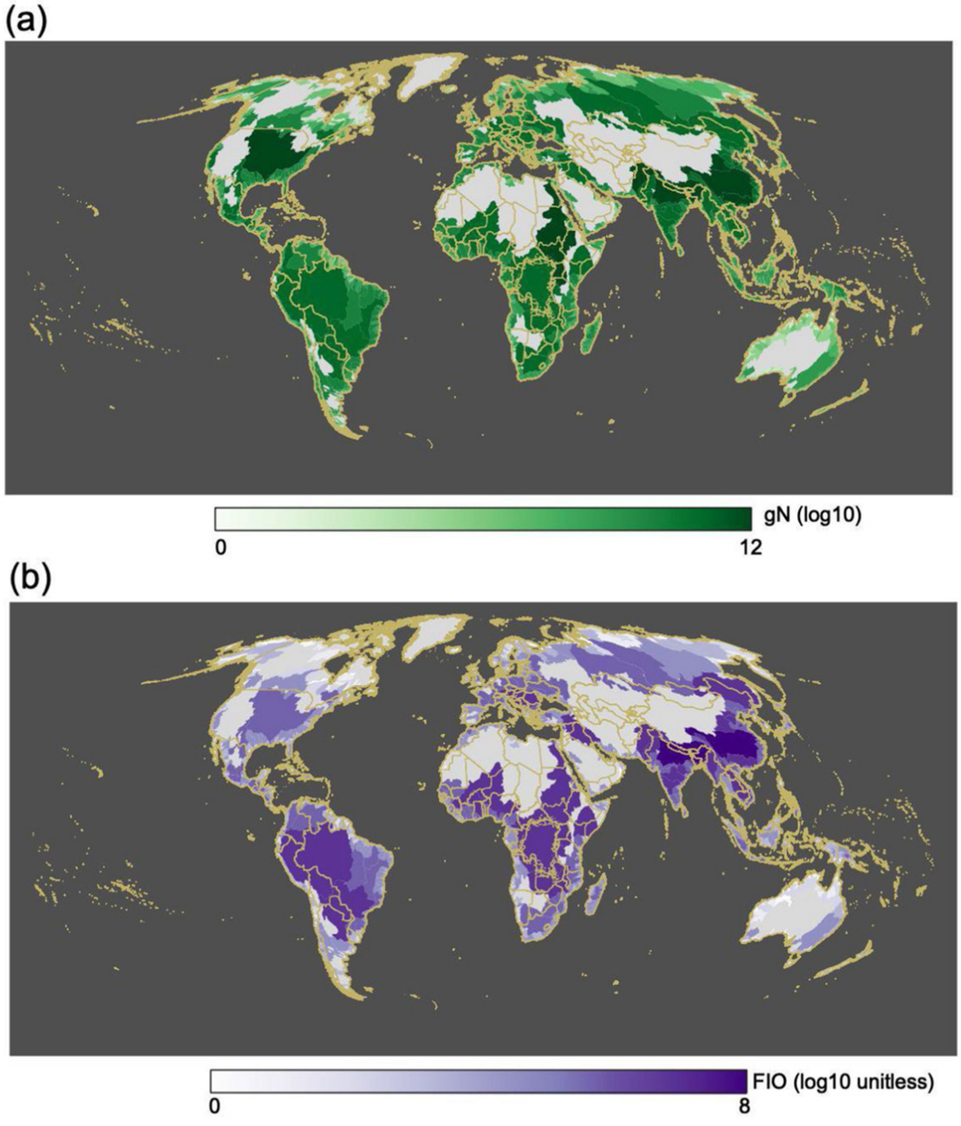

Fig. 2 (a) Maps showing terrestrial sources of wastewater (a) nitrogen and (b) fecal indicator organisms (FIO) aggregated by watersheds that empty into coastal waters. (b) Sources of both pollutants cross international boundaries (gold) requiring transboundary mitigation strategies. The disparate nitrogen and FIO loading for some major watersheds is based on how wastewater treatments remove nitrogen and FIO. For example, the Mississippi contributes far more nitrogen than FIO, reflecting the high level of tertiary treatment designed to remove pathogens but not nutrients in the United States. Data is from Tuholske *et al.*, 2021.

Although globally significant, the impacts of wastewater are spatially heterogeneous (**Fig. 2**). At any given location, complex interactions between social and environmental systems dictate wastewater impacts. For example, soil ecology, geomorphology, demographics, diet, cultural practices, and political dynamics can all affect the impacts of wastewater (Beusen *et al.*, 2016; Tuholske *et al.*, 2021). Further, wastewater inputs to coastal ecosystems often come from watersheds that have overlapping political boundaries (Tuholske *et al.*, 2021), and therefore, impacts to a coastal region cannot be easily attributed to a specific country (**Fig. 2**). Accordingly, mapping inputs from large river basins which house hundreds of millions of people and span national boundaries (e.g., the Ganges-Brahmaputra and the Danube) requires spatially-disaggregated data.

Despite the complexity of these systems and data, global wastewater models have the ability to hone in on spatially-explicit inputs related to populations, treatment types, instream retention, and often other drivers such as diet or level of economic development (Beusen *et al.*, 2016; Tuholske *et al.*, 2021). Global wastewater models produced to date measure and map nutrient (Beusen *et al.*, 2016; Tuholske *et al.*, 2021), pathogen (Kiulia *et al.*, 2015; Tuholske *et al.*, 2021; Vermeulen *et al.*, 2019), plastic (Jambeck *et al.*, 2015; Siegfried *et al.*, 2017), and pharmaceutical (Acuña *et al.*, 2020; Font *et al.*, 2019) inputs to coastal waters, with less attention paid to heavy metals and chemicals. However, only recently have models assessed the varying spatial distribution of impacts across multiple pollutant categories (Tuholske *et al.*, 2021), despite the fact that different pollutants can have contradictory impacts and divergent solutions (Tuholske *et al.*, 2021). Further, not all global wastewater models even measure discharge to oceans; rather, some only gauge surface water inputs (see Vermeulen *et al.*, 2019) without mapping the pollutant loading to coastal ecosystems.

Reducing harm to human health and well-being from wastewater has been a chief objective of international development agendas for decades, and these efforts have had a remarkable effect on improving public health. For example, from 2000 to 2017, access to safely managed sanitation services increased from 1.7 billion to 3.4 billion people worldwide, and open defecation was reduced from 1.3 billion to 673 million (UNICEF and WHO, 2020). However, the solution space for wastewater treatment can look very different when coastal ecosystems are taken into consideration. Although sewer systems are effective at removing pathogens that harm human health (UNICEF and WHO, 2020), they rarely remove pollutants that harm coastal flora and fauna. For example, wastewater treatment facilities can concentrate and emit nutrients like nitrogen and phosphorus (Mayorga *et al.*, 2010), plastics (Siegfried *et al.*, 2017), and pharmaceuticals (Acuña *et al.*, 2020; Font *et al.*, 2019). Similarly, septic systems can leach pollutants into ground and surface waters that flow into coastal environments (Herren *et al.*, 2021; Mallin and McIver, 2012). Given the rapid rise in per capita income in some countries (e.g., China, Brazil, and India) and rapid urbanization in others (e.g., Nigeria), changing diets and consumer consumption patterns present a significant challenge for coastal ecosystems, even with improved wastewater treatment in place (Tuholske *et al.*, 2021). Accordingly, policymakers worldwide must weigh limited options to protect coastal ecosystems while improving public health.

6.13.2.1 Nitrogen

Global wastewater models suggest that wastewater contributes between 4.0 and 7.2 Tg of nitrogen to coastal environments every year (Beusen *et al.*, 2016; Seitzinger *et al.*, 2010; Seitzinger and Kroeze, 1998; Tuholske *et al.*, 2021; Van Drecht *et al.*, 2009; Van Puijenbroek *et al.*, 2019). Agriculture has long been considered the primary source of nitrogen pollution in coastal and marine systems; however, according to these models, wastewater contributes roughly 45% of the amount of nitrogen that agriculture releases into the environment (Tuholske *et al.*, 2021), which highlights the substantial impact wastewater has on nitrogen loading in coastal ecosystems. Moreover, projections of wastewater nitrogen discharged to surface water, based on the Shared Socio-economic Pathways (O'Neill *et al.*, 2014), suggest that global nitrogen contributions will range from 13.5 to 17.9 Tg yr^{-1} by 2050 (Van Puijenbroek *et al.*, 2019). The range in projected nitrogen discharge reflects possible changes in protein consumption and sewage systems, in addition to plausible demographic and economic change scenarios over the next 30 years. Altogether, these models suggest that wastewater nitrogen inputs to coastal ecosystems will increase in coming decades.

A recent, high-resolution estimate of wastewater nitrogen sources suggests that in 2015, sewer systems contributed 63% (3.9 Tg) of total wastewater nitrogen inputs, septic systems contributed 5% (0.3 Tg), and direct input accounted for 32% (2.0 Tg) (Tuholske *et al.*, 2021). However, data from this model reveals that the pattern of wastewater nitrogen flows is highly spatially heterogeneous. For example, 25 watersheds contribute approximately 46% (2.8 Tg N) of global wastewater nitrogen inputs into the ocean, and a single watershed (i.e., the Chang Jiang [Yangtze] River in northern China) contributes 11% of global wastewater nitrogen (**Fig. 3**). Most of the top 25 watersheds inputting wastewater nitrogen into the ocean are located in India and China, the world's two most populated countries; however, major watersheds in various countries and most continents also rank in the top 25 (**Fig. 3**), including the Mississippi River in North America, the Niger River in Africa, the Rio Parna in South America, and the Danube in Europe.

The source of wastewater nitrogen differs greatly based on the diet composition and level of economic development of the population living within a watershed. This is well illustrated by contrasts in national-level breakdowns of wastewater input by treatment type. For example, although China emits nearly three times as much wastewater nitrogen as India does, over 70% of wastewater nitrogen from China is from treated wastewater, as compared to only 40% from India. As another example, Germany and Bangladesh both contribute about 106 Gg of wastewater nitrogen, but in Bangladesh, nearly 80% of wastewater nitrogen comes from open defecation, whereas nearly all wastewater nitrogen from Germany is treated. Further, on a per capita basis,

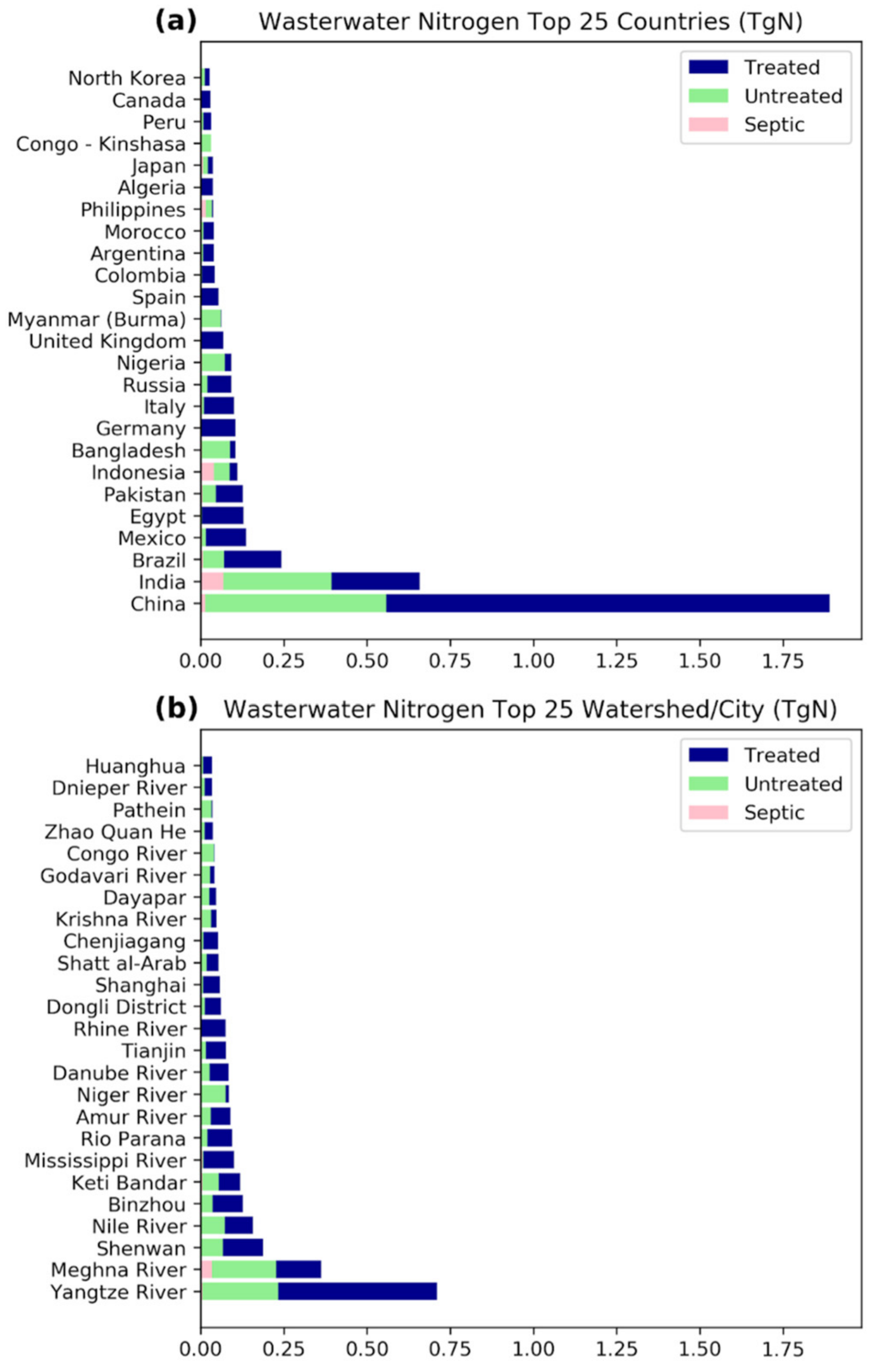

Fig. 3 (a) Stacked bar graphs showing the top 25 sources of wastewater nitrogen annually by (a) country and (b) watershed or city at the mouth of a river. (b) Color indicates the input source. Data is from Tuholske *et al.*, 2021.

Germans emit about twice as much wastewater nitrogen as Bangladesh; this reflects the high level of protein consumption in Germany compared to Bangladesh (FAO, 2017). More research is needed to understand whether the types of nitrogen species (e.g., ammonia and nitrate) being discharged into the environment differ depending on wastewater treatment level. This could be significant as different ecosystems have varying susceptibility to different species of nitrogen.

6.13.2.2 Phosphorus

The primary sources of wastewater phosphorus are human urine and laundry detergents (although the United States, European Union, and others regulate phosphorus use in detergents) (Van Puijenbroek *et al.*, 2019). Unlike nitrogen, spatially explicit data on global phosphorus flows into coastal waters, much less ecosystem impacts, is lacking. However, models show that in 2000, about 0.5 Tg of wastewater phosphorus was input into the ocean out of about 9 Tg of total phosphorus transport (Van Puijenbroek *et al.*, 2019). Further, estimates of global wastewater phosphorus discharge to surface water for 2000 ranged from 0.9 Tg yr^{-1}(Mekonnen and Hoekstra, 2018) to 1.1 Tg yr^{-1}(Van Puijenbroek *et al.*, 2019); one model estimated that figure increased to 1.5 Tg yr^{-1} for 2010 (Van Puijenbroek *et al.*, 2019). Northern China, the Nile Basin, and Central America were among the largest contributors of wastewater phosphorus to coastal waters in 2010 (Van Puijenbroek *et al.*, 2019).

6.13.2.3 Human Pathogens

Like nutrients, human pathogens are present in most inhabited coastal areas where river basins flow into oceans and coastlines (Tuholske *et al.*, 2021), and their sources are also spatially heterogeneous (**Fig. 2**). Watersheds in countries that have the least access to basic or improved sanitation tend to input the most human pathogens into coastal waters. For example, wastewater pathogen flows are greatest in sub-Saharan Africa and South Asia. In some large watersheds (e.g., the Mississippi and the Danube), there is a stark contrast between the amount of wastewater nitrogen and wastewater pathogen inputs, which reflects the effectiveness of improved sanitation at removing human pathogens, but its ineffectiveness at removing wastewater nutrients.

6.13.2.4 Contaminants of Emerging Concern

Given the range of potentially harmful substances that can find their way into wastewater, and ultimately the ocean, it is impossible to provide a full view of all contaminants of emerging concern (CECs) and their impacts. Here, we detail some that have received recent attention, while noting the significant gap in knowledge about many common pollutants. The compounds found in wastewater can occur in different concentrations ranging from nanograms to micrograms (i.e., micropollutants) to grams per liter (i.e., macro-pollutants). Emerging compounds can be broadly classified as cleaning products (e.g., disinfectants, surfactants, and bleach), personal hygiene products (e.g., UV filters, preservatives, triclosan, and microplastics), polycyclic aromatic hydrocarbons (PAHs), PCBs, pharmaceuticals (e.g., analgesics, antibiotics, anti-inflammatories, steroids, and synthetic hormones), pesticides, heavy metals, and hundreds of other compounds (Anderson *et al.*, 2004; Carballa *et al.*, 2004).

The compounds found in wastewater are continuously released into the marine environment, which makes coastal habitats susceptible to significant loading. We know that PFAS chemicals are appearing in remote parts of the ocean (Yamashita *et al.*, 2005), in plankton (Zhang *et al.*, 2019), and in the tissues of marine mammals (Gebbink *et al.*, 2016; Houde *et al.*, 2005; Palmer *et al.*, 2019), fish (Cara *et al.*, 2022; Fernandes *et al.*, 2018), and turtles (Bangma *et al.*, 2019; Wood *et al.*, 2021), which presents an emerging problem for reproductive, hormonal, and immune health of marine organisms, as well as the humans that consume them. Furthermore, with over 19,000 different prescription drug products used to improve human and animal health, pharmaceuticals are gaining greater attention from an environmental standpoint, as they are being detected in waterways and the tissues of fish. Hotspots of pharmaceutical inputs exist worldwide, across levels of economic development, and include major watersheds such as the Amazon, Mississippi, Danube, and Ganges (Acuña *et al.*, 2020). In Puget Sound, U.S.A., Meador *et al.* (2016) found that juvenile fish exposed to sewage-contaminated waters contained 42 out of 150 analytes for chemical compounds in their tissues, which could indicate why fish transiting through polluted estuaries are dying at twice the rate of non-impacted fish (Meador, 2014). Unfortunately, even the most advanced wastewater treatment plants are unable to effectively remove such novel contaminants (Khasawneh and Palaniandy, 2021; Olasupo and Suah, 2021), and these drugs have the potential to impact a range of key fitness and survival factors for different organisms including reproduction, growth, morphology, immune response, and antibiotic resistance.

In addition to pharmaceuticals, commonly consumed substances like caffeine, which originate in beverages, medicines, and chocolate, are being detected in rivers all over the world. A 2022 study found that of 1052 samples collected from 258 rivers in 104 countries, over 50% contained caffeine (Wilkinson *et al.*, 2022). Even low levels of caffeine can have negative impacts on marine species by inducing oxidative stress, neurotoxicity, metabolic activity, reproduction, development, and even mortality (Vieira *et al.*, 2022). In marine environments known to be contaminated with caffeine, Vieira *et al.* (2022) found that caffeine had accumulated in the tissues of corals, bivalves, fish, and microalgae.

Wastewater contains a host of other pollutants that are damaging to coastal ecosystems, such as plastic. In 2010, 4.8–12.7 Tg of plastic were estimated to have entered coastal waters (Jambeck *et al.*, 2015). Moreover, recent efforts to model microplastic inputs from Europe and North Africa suggest that in 2000, point sources emitted 14.4 Gg of microplastics into the North Sea, Baltic Sea, Black Sea, Mediterranean Sea, and the European river basins (all of which drain into the Atlantic Ocean) (Siegfried *et al.*, 2017). This model also suggests that over 40% of wastewater plastic inputs to coastal waters originate from synthetic polymers (e.g., household dust, as well as tires and road wear) and nearly a third come from household laundry.

6.13.2.5 Potential Synergistic Effects

The brief overview at the beginning of this chapter of the composition of wastewater pollution sheds light on the complexity of this challenge in estuarine and marine environments. Within threat mitigation frameworks, wastewater pollution is often mischaracterized as a single stressor, when it should be treated as a multi-stressor threat (Wear and Vega Thurber, 2015). Wastewater pollution is ultimately a toxic cocktail of ingredients that have the potential to synergistically increase negative impacts on vulnerable organisms and habitats when they are simultaneously faced with multiple stressors (Wear and Vega Thurber, 2015).

For example, on coral reefs, increases in sedimentation or lower light conditions can cause stress to coral colonies (Fabricius, 2005; Hodgson, 1990) and thus increase the susceptibility to pathogens introduced by wastewater pollution (e.g., white pox disease) (Sutherland *et al.*, 2010). Compounding this issue, nutrient enrichment from wastewater effluent also increases the severity of disease in corals (Bruno *et al.*, 2003). Research also suggests synergistic impacts on fish when exposed to contaminated sediments and chemicals and increased toxicity of pollutants as temperature increases (Wenger *et al.*, 2015). The potential for these sorts of synergistic impacts is likely to increase in areas with proximity to wastewater outfalls and high-density human populations (Wear and Vega Thurber, 2015). Moreover, these synergies are likely to be more common than previously appreciated given the widespread occurrence of wastewater pollution globally (Tuholske *et al.*, 2021; Wear *et al.*, 2021).

In addition to interactions and cumulative impacts from wastewater components, there is also the potential for wastewater pollution to interact with other non-pollution stressors (Wear and Vega Thurber, 2015). In particular, stressors related to climate change (e.g., sea level rise, storm surge, drought stress, and marine heat waves) all have the potential to amplify vulnerability to wastewater pollution. For example, warmer ocean temperatures can increase the virulence of pathogens and susceptibility of organisms to disease outbreaks (Bruno *et al.*, 2007; Groner *et al.*, 2021). Marine heat waves compound the effects of wastewater pollution by degrading coastal waters further via increased eutrophication and deoxygenation (Brauko *et al.*, 2020). Additionally, sea level rise and storm surge can increase erosion rates, and eutrophication can render wetlands more vulnerable to erosion because of the shift from belowground to aboveground productivity under high nutrient conditions (Deegan *et al.*, 2012). A similar result occurs when drought stress interacts with eutrophication in mangrove wetlands, leading to diebacks and die-offs when both stressors are present (Lovelock *et al.*, 2009). Finally, changes in both rainfall patterns and storm severity and frequency will result in increased flooding, which translates to more wastewater overflows from septic systems, cesspools, combined sewer overflows, and wastewater treatment plants; ultimately, this will lead to increased exposure to wastewater pollution for marine and estuarine habitats.

The following sections review what is known about the impacts of wastewater pollution on coastal and marine habitats and species. This includes the complexities and synergistic effects that result from highly variable pollutants delivered via wastewater flows into the environment. Although wastewater is a global issue, each biome has unique sensitivities and outcomes when exposed to various contaminants from wastewater.

6.13.3 Wastewater Impacts on Coral Reef Ecosystems

Coral reefs have three primary features that make them particularly susceptible to the negative effects of wastewater. First, coral reefs are dominated by invertebrates, many of which are keenly susceptible to compounds (e.g., heavy metals) and biological agents (e.g., bacteria, viruses, and eukaryotic pathogens). Second, shallow coral reefs most often, but not always, occur in oligotrophic (i.e., low nutrient) environments. In fact, many of the exciting eco-evolutionary innovations of coral reefs revolve around these nutrient and primary production limitations. Any excess nutrient inputs can alter the overall biogeochemistry of coral reef ecosystems, ultimately changing how corals settle, grow, build, and maintain the reef itself. Finally, most, but not all, coral reefs are located in areas with geomorphologies and climatologies (e.g., porous soils and sediments, shallow groundwater, and high-volume rain events) that may increase runoff or increase the inundation and retention times of wastewater. Such strong connectivity between land and sea on many fringing, patch, and barrier reefs may accelerate, exacerbate, or lengthen the effects of wastewater pollution.

Coral reefs and associated coastal systems are diverse in their hydrology (e.g., low-lying, high-permeability islands versus high islands with hard rock bases) and biological composition (e.g., hard versus soft-coral dominated); thus, making generalities about the sources and impacts of land-based pollution on coral reefs is difficult. Nevertheless, reefs are most often found in coastal areas (Spalding *et al.*, 2001), many of which have large human populations. Further, many remote high-island systems, archipelagos, and atolls now also have high-density human settlements, which proportionally contribute significant amounts of wastewater to these sensitive ecosystems. High amounts of wastewater can potentially transform coral reef systems from low-nutrient, low-disease, and typically resilient marine ecosystems, to eutrophied, diseased, and poorly adapting ecosystems. While marine heat waves induced by climate change are the biggest threat to coral reefs today, the coupling of wastewater pollution with increases in sea surface temperature can create a potentially catastrophic combination. According to Eddy *et al.* (2021), exposure to numerous stressors, including wastewater, has caused the loss of 50% of coral reefs since the 1950 s, including a 63% decrease of coral-associated biodiversity. Given the severity of the issue, identifying and mitigating local stressors like wastewater pollution may promote the natural resilience and adaptive capacity of coral reefs against climate change.

Understanding the complexity of the compounds found in wastewater and the impacts they have on coral reefs is a major research challenge. Here, we review some of the primary considerations for how and why wastewater is a serious threat to coral

Table 1 Known wastewater-derived pollutants and associated impacts on corals.

Wastewater Source Pollutant	*Documented Effects on Corals*	*References*
Freshwater	The lower salinity can cause: Bleaching, reduced growth, decreased respiration, photosynthetic rates, and mortality.	Ferrier-Pagès *et al.*, 1999; Jokiel *et al.*, 1993; Kerswell and Jones, 2003; Moberg *et al.*, 1997
Pathogens	Alter the core microbiome, Disease, mortality.	Sutherland and Ritchie, 2004; Vega Thurber *et al.*, 2014
Nutrients	Reduced growth rate, larval recruitment, bleaching, disease, low fecundity, dysbiosis.	Burkepile *et al.*, 2020; Donovan *et al.*, 2020; Stambler *et al.*, 1991; Zaneveld *et al.*, 2016
Particulates	Low light-induced bleaching, low oxygen, reduced physiological activity, disease, smoothing, and mortality.	De'ath and Fabricius, 2010; Fabricius *et al.*, 2014
Heavy Metals	Toxicity, reduced fecundity, fertilization, recruitment, settlement, bleaching, reduced physiological activity, reduced photosynthetic rate, and mortality.	Ferrier-Pagès *et al.*, 2001; Reichelt-Brushett and Harrison, 2000; Reichelt-Brushett and Michalek-Wagner, 2005
Pesticides/POPs	Toxicity, reduced photosynthetic capacity, reduced fertilization.	Glynn *et al.*, 1984; Imo *et al.*, 2008; Negri *et al.*, 2005
Microplastics	Reduced feeding capabilities, disease, tissue necrosis and bleaching, overproduction of mucus.	Huang *et al.*, 2021; Lim *et al.*, 2022; Martin *et al.*, 2019; Reichert *et al.*, 2018; Sun *et al.*, 2017
Personal Care Products	Toxicity, reduced photosynthetic capacity.	Danovaro *et al.*, 2008; Downs *et al.*, 2016; Stien *et al.*, 2020
Endocrine Disruptors	Toxicity, reduced fecundity, fertilization.	Tarrant *et al.*, 2004
Industrial and Household Cleaning Products	Toxicity, reduced tissue growth, mortality.	Shafir *et al.*, 2014
Oil and dispersant	Physiological alterations, dysbiosis, bleaching, mortality.	Fragoso ados Santos *et al.*, 2015; Häder *et al.*, 2020; Silva *et al.*, 2021

reefs, specifically highlighting how different wastewater components may affect different reef members, with a focus on the ecosystem engineers themselves (e.g., scleractinian corals). To better understand the effects of wastewater on corals, we first need to understand the effects of compounds in isolation, and then we can begin to understand the combined effects of different compounds on coral health. Each individual pollutant varies in effective dose and degree of severity when impacting coral growth and reproduction (**Table 1**).

6.13.3.1 Sources and Components of Wastewater Pollution to Coral Reefs

Corals are extremely sensitive organisms, even to small environmental changes, and wastewater-related impacts have been responsible for severe and lasting impacts on these environments (Eddy *et al.*, 2021; Pastorok and Bilyard, 1985). Wastewater is composed of 99.9% freshwater, which, if released into coastal areas, can cause hyposalinity to which corals are extremely vulnerable. According to Berkelmans *et al.* (2012), a sudden reduction in salinity can cause bleaching and death of corals, especially in shallow and coastal waters. Hyposalinity may be indirectly responsible for the decline in coral health, and thus, lead to dysbiosis and diseases like viral outbreaks (Correa *et al.*, 2016). However, identifying hyposaline stress caused by freshwater in wastewater is complex, because additional stress factors are introduced by the numerous pollutants present in wastewater (Aguilar *et al.*, 2019; Jokiel *et al.*, 1993). For example, in addition to hyposalinity, wastewater discharge can contain excess particulate matter that can increase the turbidity of seawater (Fabricius *et al.*, 2014). Turbidity can cause an increase in macroalgal cover and a reduction in the photosynthetic capacity of endosymbiotic microalgae. Additionally, the deposition of particulate material onto corals can cause disease, physiological stress, and a negative impact on coral health (De'ath and Fabricius, 2010).

6.13.3.1.1 Nutrients

Most research on the impact of wastewater on corals relates to the high levels of nutrients that enter into these environments. For example, Tuholske *et al.* (2021) found that nearly 60% of all coral reefs globally are exposed to wastewater nitrogen, while Berger *et al.* (2022) reported that approximately 80% of Mesoamerican coral reefs are exposed to pollution from excess nitrogen. As oligotrophic environments, coral reef ecosystems are sensitive to the dumping of nutrients (which usually include nitrite [NO_2^-], nitrate [NO_3^-], ammonia [NH_4], and phosphate [PO_4^{3-}]). Impacts of nutrients on corals can, directly and indirectly, affect coral reproduction (Loya *et al.*, 2004), recruitment (Szmant, 2002), and growth (D'Angelo and Wiedenmann, 2014).

Nutrient enrichment associated with wastewater discharge can cause eutrophication, a process characterized by high nutrient content in the water column. This process stimulates the growth of photosynthetic organisms in the water column, leading to decreased water clarity and decreased growth of symbiotic microalgae (i.e., Symbiodiniaceae) living in association with corals (Muscatine, 1990). As a result, the lower densities of coral-associated photosymbionts can limit carbon availability for coral growth and calcification and alter skeletal densities and homeostasis (Fabricius, 2005; Holcomb *et al.*, 2010; Langdon and Atkinson, 2005; Loya *et al.*, 2004; Stambler *et al.*, 1991). Furthermore, several reports show that an increase in nitrogen compounds in seawater intensifies bleaching during marine heat waves, but these effects are variable depending on nutrient type, concentration, and stoichiometry, as well as the exposure time, coral species, and reef environment (Donovan *et al.*, 2020; Stambler *et al.*, 1991; Vega Thurber *et al.*, 2014; Wiedenmann *et al.*, 2013). For example, nitrate can cause more severe outcomes when compared to nitrite or urea. A study done on the island of Moorea, French Polynesia, showed that nitrate increased the prevalence of bleaching of *Acropora* sp. by up to 100% and in *Pocillopora* sp. by up to 60%, but urea did not exacerbate negative effects. Further, after heat stress, coral colonies exposed to urea recovered, whereas those exposed to nitrate did not (Burkepile *et al.*, 2020). Although there are few studies on the impact of phosphate enrichment in reef environments, existing studies show that the adverse effects can be potent for corals, even though corals use phosphate concentration to accelerate growth when on reefs (Shantz and Burkepile, 2014). Excess phosphorus, in conjunction with other contaminants, can affect coral growth by altering aragonite deposition, ultimately impairing calcareous skeleton formation and making corals more fragile (Dunn *et al.*, 2012).

Excess nutrients can indirectly alter reef health by causing excessive cyanobacterial and benthic macroalgal growth. For example, cyanobacterial blooms that result from excess nutrients in oligotrophic habitats can further decrease dissolved oxygen levels and lead to hypoxia, because they increase secondary production and consume oxygen in the water column during respiration (Peña *et al.*, 2010). Cyanobacterial blooms are also associated with the microbial consortium that causes diseases such as black band disease (Meyer *et al.*, 2016; Miller *et al.*, 2011; Richardson *et al.*, 2009). Moreover, some studies have shown that cyanotoxins produced by cyanobacteria have the potential to cause histological damage to corals (Gantar *et al.*, 2009; Miller and Richardson, 2012).

The accelerated growth of benthic filamentous and macroalgae due to high nutrient content can affect corals via competitive overgrowth (Schaffelke and Klumpp, 1998; Vermeij *et al.*, 2010). When this occurs, macroalgae compete for space with corals and thus inhibit coral growth, even affecting the recruitment phase of the corals (Fabricius *et al.*, 2012). In addition, corals may suffer indirectly from shading and mechanical abrasion caused by the macroalgae (McCook *et al.*, 2001).

Excess nutrients can also lead to an increase in macro- and micro-bioeroding organisms (e.g., boring sponges), which can reduce skeletal density (Edinger *et al.*, 2000) and calcification rates (DeCarlo *et al.*, 2015), and cause a net loss of carbonate (Ward-Paige *et al.*, 2005). The sensitivity of bioeroding organisms to nutrient enrichment makes them an important bioindicator of wastewater pollution (Cooper *et al.*, 2009).

6.13.3.1.2 Pathogens and coral disease

Wastewater discharge is a direct and indirect pathway for the spread of various microorganisms, including pathogens associated with human and coral disease (Redding *et al.*, 2013; Sutherland *et al.*, 2010; Yoshioka *et al.*, 2016). The microbial community associated with coral is sensitive to environmental change, and microorganisms present in wastewater can : cause an imbalance in the coral-microorganism symbiotic association. For example, white pox coral disease, caused by the known human pathogen enterobacterium *Serratia marcescens* (Sutherland *et al.*, 2011; Sutherland and Ritchie, 2004), can cause extensive damage to coral tissue and even lead to death. In the mid-1990s, an outbreak of this disease in the Florida Keys National Marine Sanctuary spread to about 88% of the elkhorn coral (*Acropora palmata*) (Sutherland and Ritchie, 2004). This devastated the local coral ecosystem, as this species played a significant role in the structural complexity and integrity of the reef.

Previously reported coral stressors, such as high nutrient conditions and macroalgal proliferation, can also increase the incidence of disease. Higher prevalence of some diseases is strongly correlated with a high concentration of nutrients in the environment, including: black band disease, white plague type II (Kaczmarsky *et al.*, 2005), aspergillosis, yellow band disease (Bruno *et al.*, 2003), and dark spot syndrome (Vega Thurber *et al.*, 2014). Nutrient enrichment shifts the coral microbiome from mutualism to opportunism (Vega Thurber *et al.*, 2014; Zaneveld *et al.*, 2016), leading to relative changes in taxonomy and abundance of microorganisms to those that are potentially pathogenic. One group of microorganisms known as opportunists are *Vibrio* sp., which during stressful situations can rapidly proliferate in corals and induce severe bleaching, lysis, tissue necrosis, and death (Ben-Haim and Rosenberg, 2002; Munn, 2015).

Macroalgal competition with coral can also result in disease outbreaks indirectly through physical contact and the vectoring of microorganisms onto the coral. This action can lead to an increase or decrease in microbial taxa associated with corals or the establishment of new taxa (Vega Thurber *et al.*, 2012). For example, physical contact of corals with watercress algae (*Halimeda opuntia*) triggered white plague type II disease, which led to increased mortality of the corals (Nugues *et al.*, 2004).

6.13.3.1.3 Emerging chemicals, personal care products, and heavy metals

In addition to inorganic compounds and pathogenic microorganisms, hundreds of chemicals considered emerging and potentially harmful to coral reefs are also found in wastewater. Most compounds are bioactive and bioaccumulative, having a chronic and persistent environmental effect (Ballschmiter, 1996). As a result, corals are under a significant threat that can result in irreparable damage (Kroon *et al.*, 2020). Many compounds are directly toxic, while others cause cascading effects due to indirect effects or effects that require accumulation or chemical metabolism. For example, PCBs are primarily harmful due to their persistence and bioaccumulation in the environment. Over time, their toxicity increases, decreasing the abundance of endosymbiotic microalgae and reducing photosynthetic efficiency and carbon acquisition (Miao *et al.*, 2000). Although not an exhaustive list, below we highlight a few types of emerging pollutants relevant to corals. It is worth mentioning that the effects of many emerging pollutants on corals are not known, and it is likely that future studies will find direct and indirect effects of underexplored pollutants on reefs and reef organisms.

Interest in the effects of personal care products on reefs has increased significantly in recent years (Corinaldesi *et al.*, 2018; Downs *et al.*, 2016; Wijgerde *et al.*, 2020). Skin care products can contain UV filters (e.g., sunscreens and sunblocks), chemicals used to protect the skin from damaging ultraviolet radiation of sunlight (UV). The active ingredients in UV filters absorb radiation, then either dissipate dissolved energy through photophysical and photochemical pathways (Serpone *et al.*, 2007) or reflect UV radiation by scattering the UV photons (Salvador and Chisvert, 2017; Serpone *et al.*, 2007). This link to photophysiology is the primary concern for coral reefs. Although research in this area has been mixed, clear evidence (reviewed by Miller *et al.*, 2021) shows that UV filters sensitize cnidarians to photoinhibition and bleaching (Fel *et al.*, 2019; Vuckovic *et al.*, 2022). The magnitude of UV filter pollution was assessed in a study in Mexico, in which a single lagoonal site was estimated to experience 240 tonnes of sunscreens every year (Casas-Beltran *et al.*, 2020), and other data suggest that at least 10% of reefs are likely to experience coral bleaching due to pollution from UV filters (Downs *et al.*, 2016; Stien *et al.*, 2020). These data have prompted several regional bans of sunscreens and sunblocks that contain specific active compounds, such as oxybenzone.

Coral stress responses to heavy metals are well reported in the literature and include ontogenic and adult toxic effects. For example, heavy metals can directly affect the reproductive process, by inhibiting coral fertilization (Reichelt-Brushett and Michalek-Wagner, 2005), reducing larval settlement, and decreasing rates of polyp division (Ferrier-Pagès *et al.*, 2001; Reichelt-Brushett and Harrison, 2000). In addition, heavy metals can cause physiological stress, resulting in the bleaching of corals and a reduction in coral calcification rates due to changes in the population of endosymbiotic photosynthetic microalgae.

6.13.3.1.4 Microplastics

Another active area of research is the effects of microplastic polymers on coral health. Microplastics are present in wastewater via disposal of microfibers and polyesters in textiles or the fragmentation of larger plastics such as polyethylene terephthalate (PET) (Liu *et al.*, 2019a; Šaravanja *et al.*, 2022). Their small size and compositional variation make them particularly challenging to work with; microplastic fragments can range in size from 0.1 μm to 5 mm, while nanoplastics range from 0.001 to 0.1 μm. Of particular concern is that these size ranges are similar to the plankton targeted by filter-feeding organisms such as corals. Thus, their size facilitates ingestion by and accumulation in corals (Fendall and Sewell, 2009; Hall *et al.*, 2015; Lim *et al.*, 2022). Studies show that microplastics can affect coral growth rate, reproduction, and physiology which, in turn, causes tissue necrosis, bleaching, and excessive mucus production (Huang *et al.*, 2021; Lim *et al.*, 2022). One study also found that the abundance of microplastics was relatively greater on the surfaces of corals than inside the skeletons (Martin *et al.*, 2019). Along with their direct effects, microplastics can serve as vectors for other pollutants through the physical transport of contaminants such as heavy metals, PAHs, and bisphenol A (BPA) (Barboza *et al.*, 2018; Verla *et al.*, 2019), magnifying their negative impacts on reefs (see below).

6.13.3.1.5 Polyaromatic and other hydrocarbons

Polyaromatic and other hydrocarbons are pervasive and can be long-lasting in the oceans. These pollutants gained worldwide visibility after one of the largest marine disasters in modern history, the Deepwater Horizon oil spill, where approximately 206 million barrels of oil were released into the Gulf of Mexico. While hydrocarbons naturally occur, they are pollutants because they are highly toxic, carcinogenic, and mutagenic, with low volatility (Abdel-Shafy and Mansour, 2016; Varjani and Upasani, 2017). Most studies concerning PAHs/hydrocarbons and corals revolve around oil spills, but these are also common contaminants in runoff and in wastewater (Hsu *et al.*, 2016). These pollutants can affect the settlement and recruitment of coral larvae (Hartmann *et al.*, 2015), induce DNA damage (Fu *et al.*, 2012), and precipitate high rates of coral mortality at sufficiently high concentrations. Furthermore, oil fractions can cause extensive tissue damage due to the accumulation of the water accommodated fraction and changes in the coral microbiome, causing dysbiosis in the holobiont coral (White *et al.*, 2012).

6.13.3.1.6 Endocrine disruptors and coral reproduction

Some compounds in wastewater can interfere with the functions of the endocrine system, causing adverse effects on organisms, such as infertility, feminization, and decreased reproduction rate (Carson, 2002; Gonsioroski *et al.*, 2020; Sumpter, 2005). These endocrine disrupting compounds include natural and synthetic hormones, PAHs, PCBs, phthalates, herbicides, and BPA (Petrovic *et al.*, 2004; Schug *et al.*, 2011). Indeed, when colonies of rice coral (*Montipora capitata*) and finger coral (*Porites compressa*) were exposed to the effects of 17β-estradiol and estrone, the number of gametes and sperm from the rice coral decreased, and finger coral had reduced skeletal growth rates (Klančič *et al.*, 2022; Tarrant *et al.*, 2004). BPA and herbicides such as Diuron and Irgarol 105 are found in graywater, or released into the environment through polycarbonate plastics, epoxy resins, and herbicides. BPA can penetrate coral tissue and dramatically reduce the photochemical efficiency of endosymbiotic microalgae. This reduction in efficiency can result in an avalanche of adverse effects on the coral. Furthermore, herbicides can directly affect photosynthetic capacity, culminating in coral bleaching (Glynn *et al.*, 1984; Negri *et al.*, 2005).

Chemicals we use daily, such as surfactants (e.g., detergents and dispersants) can also cause disintegration of some biological and abiotic components of coral reefs. It is estimated that the world production of surfactants in 2006 was 12.5 million tonnes (Ivanković and Hrenović, 2010), but production has almost certainly grown since then. In coral reefs, studies on the effects of industrial and domestic surfactants are still nascent and more studies should be conducted. The surfactants nonylphenol ethoxylated (found in industrial detergents, such as laundry) and linear alkyl benzene sulfonates used in household products resulted in high mortality and reduced tissue growth in different coral species (e.g., hood coral [*Stylophora pistillata*] and lace coral [*Pocillopora damicornis*]) (Shafir *et al.*, 2014). In addition to these effects, surfactants are also considered endocrine disruptors (Miles-Richardson *et al.*, 1999). Some drugs, such as antibiotics, estrogenic hormones, and anti-inflammatories, are also toxic to

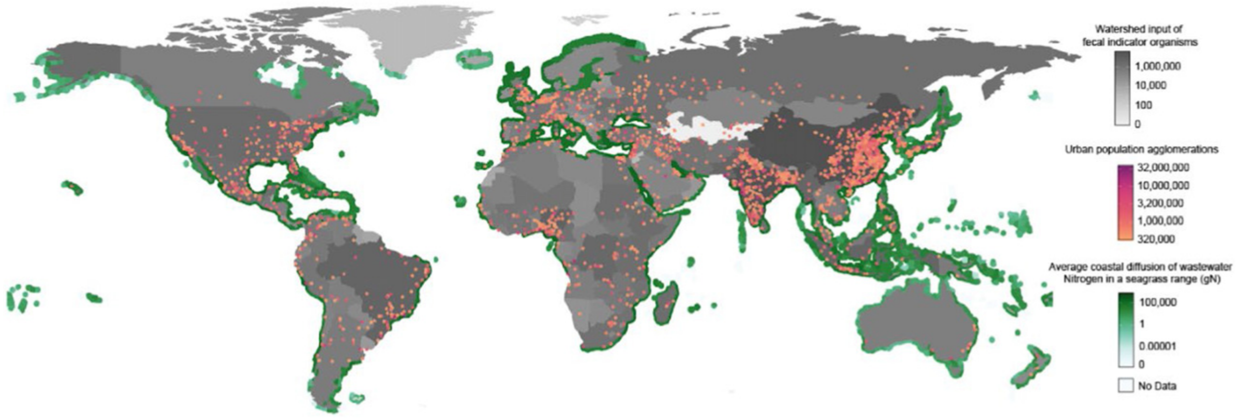

Fig. 4 Map showing the watershed input of fecal indicator bacteria to coastal oceans (by country), locations of urban population agglomerations, and global coastal diffusion of wastewater nitrogen in seagrass ecosystem ranges. World urbanization prospect data from the United Nations Department of Economic and Social Affairs, Population Division (United Nations, 2018). Seagrass species distributions from the International Union for Conservation of Nature (IUCN) Red List of Threatened Species (IUCN, 2019). Urban agglomerations are defined as populations greater than 300,000 people. Data on total wastewater nitrogen and watershed input of fecal indicator organisms from Tuholske *et al.*, 2021.

corals and considered endocrine disruptors. In corals, antibiotics have the ability to alter the coral microbiome and cause microbial dysbiosis, which can lead to death (Connelly *et al.*, 2022; Dunphy *et al.*, 2021).

6.13.4 Wastewater Impacts on Seagrass Ecosystems

Seagrasses are the most widespread coastal ecosystems on the planet, covering an area of over 300,000 km^2; they are found in 159 countries and on every continent except for Antarctica (UNEP, 2020). Seagrasses are a polyphyletic group of submerged marine flowering plants with 72 species that evolved in multiple instances from terrestrial plants (Olsen *et al.*, 2016). Seagrasses are one of the most valuable coastal and marine ecosystems, providing a range of critical environmental, economic, and social benefits (Unsworth *et al.*, 2022). In addition to supporting thousands of species of marine organisms and 20% of the largest fisheries (Unsworth *et al.*, 2019), seagrass ecosystems provide services that combat climate change effects via carbon sequestration, acidification buffering, disease control, and protection from extreme weather events (Unsworth *et al.*, 2022).

Seagrass ecosystems predominantly occupy shallow coastal waters (Coles *et al.*, 2009), and this proximity to large coastal populations makes seagrass ecosystems especially vulnerable to human impacts. Seagrass coverage has declined globally by 110 km^2 yr^{-1} since 1980 with an overall loss of 30% since the late 19th century (UNEP, 2020; Waycott et al., 2009). Over one-third of global urban agglomerations (i.e., populations of at least 300,000 people) occur within 50 km of a seagrass ecosystem (**Fig. 4**), with an estimated 88% of seagrass ecosystems exposed to nitrogen from wastewater (Tuholske *et al.*, 2021). As urbanization of coastal areas increases, seagrass ecosystems face pressures from co-occurring wastewater pollutant runoff, mechanical damage, and climate change impacts.

Here, we outline the state of knowledge of the effects of the constituents of wastewater on seagrass ecosystems, the importance of mitigating wastewater pollutants for seagrass recovery and conservation, and the role of resilient seagrass ecosystems in reducing wastewater-associated impacts on human and ocean health.

6.13.4.1 Nutrients, Metals, and Microbes

Wastewater pollution is a global driver of seagrass ecosystem degradation, with impacts ranging from microscale mechanisms to ecosystem-wide degradation (**Table 2**). Not only can wastewater have a direct impact on seagrass ecosystems located adjacent to outfalls, but also can have wide-ranging impacts via river transport (Cabaço *et al.*, 2008; Thangaradjou *et al.*, 2014). For example, the area surrounding a sewage treatment outfall near Adelaide, South Australia declined from 85% seagrass cover to completely denuded following 15 years of operation (Bryars and Neverauskas, 2004).

Nutrients and metals are two of the most comprehensively studied wastewater impacts on seagrass ecosystems (**Table 2**). Nutrients are essential for seagrass growth, but concentrated levels of nutrients in wastewater cause eutrophication of seagrass ecosystems. Eutrophication occurs when nutrients, particularly nitrogen, encourage the overgrowth of algae (Nelson, 2017). On seagrass blades, epiphytic algae block light transmission and reduce photosynthesis, ultimately leading to plant death (Lapointe *et al.*, 2015; Mabrouk *et al.*, 2014; Nelson, 2017). Metals can also negatively impact seagrasses when they accumulate within seagrass tissue (Lin *et al.*, 2016). Direct toxic effects on seagrass include impaired cellular function, oxidative stress, and altered gene expression (**Table 2**). Moreover, metals persist within the marine environment, leading to cumulative effects on seagrass and associated organisms over time.

Table 2 Impacts of wastewater constituents on seagrass ecosystems.

Constituent	*Impact*	*Mechanism*	*Reference*
Nutrients			
Mixed Nutrients	Eutrophication	Nutrient enrichment leads to overgrowth of epiphytes, which smothers seagrass plants and reduces photosynthesis	Lapointe *et al.*, 2015; Nelson, 2017
	Food web alteration	Modified food web structure	Cui *et al.*, 2021
	Growth alteration	Increased number of leaves per shoot	Wang *et al.*, 2020b
	Microbiome alteration	Increased sulfur- and nitrogen-cycling bacteria	Wang *et al.*, 2020b
Nitrate	Disease susceptibility	Increased susceptibility to Wasting Disease by infection with *Labyrinthula zosterae*	Hughes *et al.*, 2018
Ammonium	Growth alteration	Decreased leaf elongation rate and photosynthesis, increased root activity	Wang *et al.*, 2021
	Altered nutrient distribution	Increased ammonium in root and leaves, decreased total nitrogen in leaves/ increased in root	Wang *et al.*, 2021
	Altered enzyme activity	Alterations of genes relating to glutamine and glutamate	Wang *et al.*, 2021
	Growth alteration	Decreased patch expansion	Govers *et al.*, 2014
Urea	Growth alteration	Decreased seagrass morphology indices	Li *et al.*, 2019
	Altered nutrient distribution	Carbon and nitrogen imbalance in plant tissues	Li *et al.*, 2019
Sulfides	Toxicity	Oxidative stress, decreased chlorophyll	Parveen *et al.*, 2017
	Toxicity	Reduction is qualitative plant metrics	Pedersen and Kristensen, 2015
	Toxicity	Decreased survival, decreased patch expansion	Govers *et al.*, 2014
	Toxicity	Seedling mortality	Dooley *et al.*, 2013
	Growth alteration	Decreased production of new leaves per shoot	García *et al.*, 2012
Microorganisms			
Enterococcus	Amplification	Colonizes seagrass biofilms, where it replicates	Ferguson *et al.*, 2016
Metals			
Mixed heavy metals	Toxicity	Antioxidant response	Bertini *et al.*, 2019
Copper	Toxicity	Impaired chloroplast function	Mohammadi *et al.*, 2019
	Toxicity	Oxidative stress	Greco *et al.*, 2019; Zheng *et al.*, 2018
	Toxicity	Phytotoxicity/reduced chlorophyll, leaf necrosis	Zheng *et al.*, 2018
	Toxicity	Reduced photosynthesis, epigenetic modifications	Greco *et al.*, 2019
	Toxicity	Seedling mortality, altered chlorophyll content	Zhao *et al.*, 2016
	Toxicity	Microtubule disturbance and cell death	Malea *et al.*, 2013b
Cadmium	Toxicity	Oxidative stress	Greco *et al.*, 2019; Zheng *et al.*, 2018
	Toxicity	Microtubule depolymerization leading to cell damage and death	Malea *et al.*, 2013a, Malea *et al.*, 2014b
Zinc	Toxicity	Oxidative stress	Zheng *et al.*, 2018
Silver nanoparticles	Toxicity	Alterations of the cytoskeleton, endoplasmic reticulum, ultrastructure, photosystem II function, oxidative stress, cell viability, and leaf, rhizome and root elongation	Mylona *et al.*, 2020b
Titanium dioxide	Toxicity	Increased hydrogen peroxide, impaired actin filaments and endoplasmic reticulum	Mylona *et al.*, 2020a
Arsenic	Microbiome alteration	Increased iron-cycling bacteria	Martin *et al.*, 2022

Table 2 Continued

Constituent	*Impact*	*Mechanism*	*Reference*
Lead	Toxicity	Cellular microtubule damage and death	Malea *et al.*, 2014a
	Leaf mortality	Decreased growth rate, increased leaf necrosis	Nielsen *et al.*, 2017
Iron	Toxicity	Increased seedling mortality	Wang *et al.*, 2017
	Microbiome alteration	Increased iron-cycling bacteria	Martin *et al.*, 2022
Mercury	Biomagnification	Biomagnification	Coelho *et al.*, 2013
Nickel	Toxicity	Microtubule disturbance and cell death	Malea *et al.*, 2013b
Chromium	Toxicity	Microtubule disturbance and cell death	Malea *et al.*, 2013b
Plastics			
Macroplastic	Growth alteration	Increased spacing between shoots	Menicagli *et al.*, 2021
	Species competition	Reduced ability to compete with invasive algae	Menicagli *et al.*, 2021
Microplastic	Accumulation	Encrusting on leaves leading to biomagnification in ecosystem	Goss *et al.*, 2018; Sanchez-Vidal *et al.*, 2021
Chemicals			
Bisphenol A (BPA)	Toxicity	Cellular loss of chlorophyll auto-fluorescence and oxidative stress	Adamakis *et al.*, 2021
	Growth alteration	Inhibition of blade elongation	Malea *et al.*, 2020
Polycyclic Aromatic Hydrocarbons (PAH)	Microbiome alteration	Increase in PAH-degrading microorganisms	Ahmad *et al.*, 2021a; Ahmad *et al.*, 2021b
	Microbiome alteration	Shift in total microbial community	Ling *et al.*, 2015
Antimicrobial Compounds	Microbiome alteration	Increase in drug-resistant bacteria	Jebara *et al.*, 2022
Freshwater			
Reduced salinity	Mortality	Seagrass mortality, especially in smaller patches	Stipek *et al.*, 2020
	Growth alteration	Slowed leaf growth	Milbrandt and Siwicke, 2016

As primary producers and foundation species, wastewater impacts on seagrass plants affect all levels of the food web. For instance, eutrophication causes a modified food web structure and shifts grazers towards algal food sources (Cui, 2021). Conversely, sea otters promote a trophic cascade that encourages seagrass growth under nutrient enrichment (Hughes *et al.*, 2016). Additionally, concentration of pollutants within seagrass plants and subsequent ingestion by grazers leads to biomagnification of wastewater constituents such as metals and microplastics (Coelho *et al.*, 2013; Goss *et al.*, 2018; Sanchez-Vidal *et al.*, 2021).

Despite the importance of the seagrass epiphyte community for proper plant and ecosystem function, little research has explored specific chemical and microbial pressures on seagrass from wastewater. However, nearly all constituents of wastewater have been linked to the alteration of the seagrass microbiome (**Table 2**). For instance, bacteria of the genus *Enterococcus*, a common constituent of wastewater and indicator of fecal pollution, has been shown to colonize and replicate on the surface of seagrass plants (Ferguson *et al.*, 2016).

6.13.4.2 Wastewater Mitigation and its Role In Seagrass Conservation and Recovery

Marine protected areas (MPAs) are one of the most common strategies to protect coastal ecosystems; however, only 26% of seagrass meadows are located inside an MPA (as compared to 40% of coral reefs and 43% of mangroves). Wastewater pollution will likely undermine permeable spatial boundaries from diffuse coastal watershed inputs. For example, Jones *et al.* (2018) found markers of wastewater pollution across all MPA sites with seagrass ecosystems.

Restoration by natural recolonization or planting of seagrass can be effective for restoring ecosystem services and counteracting biodiversity loss. The average cost to actively restore 1 ha (0.01 km^2) of seagrass is over USD $400,000 (Bayraktarov *et al.*, 2016), highlighting the value of natural restoration by improving water quality. Despite a general global trend of seagrass ecosystem loss, there are a range of examples that indicate wastewater pollution mitigation can lead to recovery of seagrass ecosystems. For example, proper wastewater treatment can reduce nutrient and pollutant inputs into seagrass ecosystems and reverse declines in seagrass cover (Johansson and Lewis, 1992; Johansson and Ries, 1996; Pergent-Martini *et al.*, 2002, Tomasko *et al.*, 2018). Sewage outfall diversion and decommissioning can also promote seagrass recovery. For example, eight years after decommissioning a sewage treatment outfall, a formally cleared 2 ha (0.02 km^2) site regained 23% seagrass cover (Bryars and Neverauskas, 2004).

6.13.4.3 Importance of Seagrass for Detecting Wastewater Impacts

Seagrass ecosystems serve as a historical record for constituents of wastewater. Accumulation of metals and nutrient isotopes in seagrass leaves reflects local pollution over a moderate timescale. For instance, as tourism and related wastewater discharge increased in Quintana Roo, Mexico, levels of ^{15}N, a marker of anthropogenic pollution, increased in the tissue of local turtle-grass (*Thalassia testudinum*) (Sánchez *et al.*, 2013). Therefore, routine sampling of seagrass tissue can serve as an early warning system for wastewater pollution.

6.13.5 Wastewater Impacts on Mangrove Ecosystems

Mangroves are a community of woody plants from a wide range of plant families. They share traits of tolerance to saline and brackish water and inundation by tidal water, although tolerance varies among species (Duke *et al.*, 1998). Mangroves are found in the intertidal zone of low energy and sandy to muddy shorelines of the tropics and subtropics, often forming extensive stands in estuaries. Mangroves provide habitat for a wide range of vertebrate and invertebrate fauna that can have affiliations with other marine (e.g., coral reefs and seagrass) or terrestrial (e.g., lowland forests) environments, because fauna move in and out of the habitat at different life-history phases (Nagelkerken *et al.*, 2008; Sievers *et al.*, 2019). For example, mud crabs (*Scylla serrata*), which are an important fishery species, are juveniles in the ocean, but spend their adult phase in mangrove habitats.

In addition to supporting fisheries and biodiversity, mangroves provide a range of other ecosystem services. They provide coastal protection from storms and floods, and they accumulate carbon in biomass and soils, making them important for global climate regulation (Barbier *et al.*, 2011). High carbon stocks and rates of carbon sequestration in mangroves have led to them being considered as blue carbon ecosystems, which has stimulated conservation and restoration activities (Lovelock and Duarte, 2019).

Global mangrove cover is approximately 137,000 km^2 (Giri *et al.*, 2011). This ecosystem has been degraded and converted, mainly for production of commodities such as rice and shrimp aquaculture (Goldberg *et al.*, 2020). While rates of conversion and loss have slowed over the past decade, the impacts of climate change are increasing, with evidence of shoreline retreat in many locations (Goldberg *et al.*, 2020).

Of the different types of coastal ecosystems, mangroves are one of the few that still persist within urban settings (Mazor *et al.*, 2021) and are thus likely to be strongly influenced by wastewater. Moreover, mangroves have been intentionally used as sites for wastewater treatment (Ouyang and Gou, 2016) because of their ability to retain suspended solids and remove nutrients via plant growth and microbial processes. While constructed wetlands (e.g., those built to take up solids and nutrients from aquaculture) are beneficial for reducing pollution delivered to waterways and adjacent ecosystems, wastewater impacts on mangroves can have negative influences both on mangrove flora and fauna, and on the people who use the resources that are within mangroves (Crona *et al.*, 2009). Because mangroves are intensively used in many nations for harvesting of wood and fish (see Aye *et al.*, 2019), wastewater pollution can be particularly detrimental to dependent communities who are exposed to contaminants while visiting the mangroves and consuming seafood products (Crona *et al.*, 2009).

6.13.5.1 Nutrients

Nutrient enrichment from wastewater usually enhances mangrove tree growth (Clough *et al.*, 1983; Erftemeijer *et al.*, 2021) by alleviating nutrient limitations (Reef *et al.*, 2010). Research using experimental nutrient additions (albeit in different nutrient forms to those that occur in wastewater) indicates that mangrove trees that are usually highly efficient at resorbing nutrients from senescing leaves experience a reduction in efficiency, and therefore release nutrients back into the environment in leaf detritus (Feller *et al.*, 1999). Moreover, in some sites, enhanced growth rates with nutrient enrichment can lead to increases in soil surface elevation, which is an important process by which mangroves adapt to sea level rise (McKee *et al.*, 2007). Despite the positive influence of wastewater nutrients on plant growth, nutrient enrichment experiments suggest that rapid tree growth rates may be associated with greater vulnerability to extreme climatic events, like drought and intense storms (Feller *et al.*, 2015; Lovelock *et al.*, 2009). High nutrient levels result in plants investing proportionally greater biomass in canopies, rather than heavily lignified stems and large root systems; this reduction in belowground biomass limits tolerance to extreme events and may render wastewater-impacted mangroves more vulnerable to climate change. Moreover, the effects of wastewater on trees can be persistent. For example, in Brisbane, Australia, trees displayed the isotopic signature of sewage nitrogen two years after the improvement of sewage treatment plants, compared to crab fauna that did not; this indicates that trees may continue to use sewage nitrogen stored in sediments even after conditions have improved (Pitt *et al.*, 2009).

Nutrients in wastewater can modify microbial communities (Craig *et al.*, 2021), and enhance the growth of invertebrates (Penha-Lopes *et al.*, 2011) and macroalgae on aboveground roots, stems, and sediments (Melville and Pulkownik, 2006). In general, many mangrove-associated taxa exposed to elevated nutrients exhibit patterns of enhanced biomass, at the expense of declining diversity, thus reducing functional redundancy in mangrove ecosystems and potentially increasing their vulnerability to other stressors. Experimental evidence suggests that nutrient enrichment reduces the diversity and function of microbial communities (Craig *et al.*, 2021), but does not affect mangrove folivory (i.e., consumption of leaves by insects and crabs) (Feller *et al.*, 2013). Behavioral changes in crabs were observed with wastewater inputs that were linked to high levels of food resources. The depth of crab burrows, and hence oxygenation of sediments, was also reduced (Bartolini *et al.*, 2009). Furthermore, excessive organic inputs can stimulate microbial growth that depletes oxygen levels in water and sediments, resulting in suboptimal conditions for growth of most taxa, and even reduced density of crabs (Theuerkauff *et al.*, 2020), although species tolerances are likely to vary (Kon *et al.*, 2022).

6.13.5.2 Metals

Wastewater not only contains nutrients and organic matter, but also a range of metals and other chemical and microbial contaminants that accumulate in mangrove plant tissues and sediments (Araújo *et al.*, 2021; Robin *et al.*, 2021; Tam and Wong, 2000) at concentrations that often exceed human health thresholds (Branoff, 2018). Although bioaccumulation of metals differs among mangrove species (Robin *et al.*, 2021), a recent study of *Avicennia* and *Rhizophora* stands found that metal accumulation was enhanced by higher soil salinity (Bourgeois *et al.*, 2020). Adverse effects of metals on mangrove tree growth have been observed (Nguyen *et al.*, 2020), but there are few data to develop quantitative thresholds for vital processes (Yan *et al.*, 2017). Mangrove epiphytic root algal communities are sensitive to metal concentrations (Melville and Pulkownik, 2006), and heavy metals adversely influence mangrove macrofauna and can accumulate in animals harvested for seafood (e.g., mangrove oysters, snails, and crabs) (Arumugam *et al.*, 2018). However, a study on estuarine fish in polluted estuaries in Southeast Asia found that metal levels in fish were generally safe for human consumption (Pandion *et al.*, 2022).

While mangrove sediments and plant biomass store heavy metals, they also release accumulated metals into the water column and can therefore contribute to the export of metals from wastewater into adjacent marine environments. In Vietnam, metal export from mangroves was associated with decomposition of sediment organic matter (Thanh-Nho *et al.*, 2019), suggesting that nutrient enrichment or mangrove damage during extreme events (which are more likely with wastewater pollution) could enhance metal fluxes from mangroves to the water column and adjacent environments. Finally, wastewater discharge is also associated with freshwater discharge. Mangrove trees generally grow more rapidly in low, and even fluctuating, salinity conditions (i.e., salinities less than seawater) (Wang *et al.*, 2020a), but fluctuating levels of salinity can be detrimental to many invertebrates (Rivera-Ingraham and Lignot, 2017).

6.13.6 Wastewater Impacts on Salt Marsh Ecosystems

Salt marshes are among the most productive systems in the world and generate vital services for humans, including erosion protection (Gedan *et al.*, 2009; Silliman *et al.*, 2019), nutrient filtration, carbon storage (Mcleod *et al.*, 2011), and fisheries enhancement (Nagelkerken *et al.*, 2008). Salt marshes are biogenic communities, and one or a few foundation species plants, such as smooth cordgrass (*Spartina alterniflora*), build their structure (Silliman, 2014). Decades of research have shown that the growth of foundational marsh plants is controlled by both nitrogen limitation (Mendelssohn *et al.*, 1981) and grazers (Silliman and Bertness, 2004), and that physical factors and competition interact to generate the striking plant zonation in these systems (Bertness, 1991). Managers and ecologists have long viewed marshes as systems that are relatively resilient and thus not controlled by human impacts (Valiela *et al.*, 1975); however, recent research has shown that salt marshes are under global threat from a range of human effects, including nutrient enrichment and contaminant build-up from wastewater pollution (Gedan *et al.*, 2009; Silliman *et al.*, 2009), as well as effects of climate change (sea-level rise and storm surges) and hydrological modifications that alter freshwater inputs, residence times, and tidal activity that delivers sediments to marsh surfaces resulting in vertical accretion.

6.13.6.1 Nutrient Enrichment and Salt Marsh Vulnerability

Traditionally, salt marshes were thought to be wastewater-resistant (Breteler *et al.*, 1981), because (Valiela *et al.*, 1975): (1) they are nitrogen-limited and wastewater is rich in nitrogen, so wastewater should support marsh plant growth; (2) salt marsh soils have high bacterial diversity and steep gradients in redox potential, so marshes are efficient biogeochemical-processing units, which should be able to rapidly transform wastewater organics into plant and animal biomass and inorganic gases; and, (3) their soils lack oxygen, so many of the heavy metals in wastewater can be transformed into less toxic forms. In fact, short-term studies of sludge application to salt marshes in the 1970s and 1980s suggested that salt marshes would benefit from sewage application (Chalmers, 1979; Hanson, 1977; Valiela *et al.*, 1975). These studies were motivated, in large part, to test the hopeful idea that salt marshes could act as natural wastewater treatment plants and thus be included in an area's wastewater treatment plans. For example, results showed that aboveground biomass of marsh plants increased in response to sewage sludge addition (Valiela *et al.*, 1975), suggesting that marsh plants could potentially benefit or not be affected by wastewater pollution and should thus, at a minimum, be conserved to help uptake nutrient pollution and potentially be used explicitly in wastewater treatment plans.

Research in recent decades has challenged the notion that salt marshes are wastewater resistant and revealed that there are severe ramifications of long-term nutrient enrichment on salt marsh ecosystems (Bertness *et al.*, 2002; Deegan *et al.*, 2012). Warnings that nutrient enrichment could have deleterious effects on salt marshes first emerged from plant competition studies in New England. When researchers added nitrogen fertilizer to the borders between marsh plant zones, lower marsh plants became dominant and overgrew upper marsh plants because nitrogen fertilization reversed the competitive hierarchy (Levine *et al.*, 1998). Indeed, two other unexpected interactions also occurred in fertilized plots: (1) increased success of an invasive plant species, common reed (*Phragmites australis*) (Minchinton and Bertness, 2003); and, (2) increased grazing pressure by insects on native plants (Sala *et al.*, 2008). Large-scale, follow-up, surveys found that these results were spatially general and that development and poor management of nonpoint pollution were contributing to changes in salt marsh composition (Bertness *et al.*, 2002; Silliman and Bertness, 2004). Specifically, these studies found that shoreline development increased nutrient input to marshes and that increased availability of nutrients led to increased grazing on native plants, wholesale takeover of marshes by common reed, and subsequent declines in native plant diversity.

More recently, long-term additions of inorganic nitrogen in large quantities to salt marsh creeks led to both expected and unexpected outcomes (Deegan *et al.*, 2012). As was predicted from past studies, increasing nitrogen availability led to increased aboveground biomass in plants; however, fertilization had the opposite effect on belowground plant biomass. Root systems of plants in fertilized creeks experience a reduction in fine roots and become shallower and less dense. Creek banks in fertilized creeks were no longer held together by abundant plant roots and began to calve and experienced elevated erosion. A similar negative effect of fertilization on creek bank integrity has been documented in West Coast marshes of the United States, where nutrients fuel growth of algal mats that kill creek bank plants (Wasson *et al.*, 2017). Graham and Mendelssohn (2014), however, conducted research in an oligohaline marsh and found contrasting results to Deegan *et al.* (2012). Specifically, after 13 years of fertilization in the marsh interior, Graham and Mendelssohn (2014) found that nutrient enrichment did not destabilize marsh habitat. These contrasting results could be driven by the two experiments being conducted in different areas of the marsh (interior versus edge) and/or that there is considerable variation in marsh responses to nutrient addition; it's evident that more studies are needed to reveal the underlying mechanism.

Salt marsh nutrient enrichment studies highlight that wastewater pollution is very likely to have strong negative impacts on marsh structure and function. Instead of marshes acting as natural wastewater treatment plants that benefit from the growth-limiting elements of wastewater (e.g., nitrogen), marsh structure is very likely to degrade, and nutrient enriched marshes will likely experience increased erosion, intensified top-down control, increased plant invasions, lower plant diversity, and reduced ability to keep pace with sea level rise. Below, we summarize what is known from studies that directly examine the impacts of wastewater pollution (both nutrients and contaminants) on salt marshes.

6.13.6.2 Exposure Extent and Consequences for Salt Marshes

While many investigations have examined the effects of nutrient addition on salt marshes, far fewer have directly tested for the effects of wastewater pollution or looked at how the numerous contaminants found in wastewater (e.g., pharmaceuticals, hormones, heavy metals, and microplastics) can affect salt marsh structure and function. Moreover, marine ecologists have long thought wastewater pollution was a relatively localized threat (Tuholske *et al.*, 2021), but recent studies have shown that a large proportion of salt marshes around the world experience high levels of wastewater exposure (Deegan *et al.*, 2012; Wear *et al.*, 2021). For example, Wear *et al.* (2021) used predicted diclofenac (DCL) concentrations as an indicator for overall risk of wastewater pollution exposure and categorized salt marshes by the severity of the wastewater inundation threat. Their analysis found that approximately 30.9% of salt marshes worldwide exist in areas of high (7.5–10 ng L^{-1}) or very high (>10 ng L^{-1}) DCL concentration. Furthermore, because salt marshes are predominantly located along mid-latitude coastlines, which coincide with a large portion of cities around the world, they are likely to have higher levels of DCL, and thus wastewater pollution (Wear *et al.*, 2021).

The relatively recent and consistent findings that salt marshes are highly degraded when nutrient enrichment occurs and that they are commonly, rather than rarely, bathed in high levels of wastewater, highlights the importance of studies that have looked directly at wastewater impacts on salt marshes. Specifically, even though wastewater pollution in marshes can generate short-term benefits to aboveground plant growth, in the long term, marshes that experience sustained nutrient addition decrease in elevation and erode more, both effects that greatly undermine marsh resilience to sea level rise (Deegan *et al.*, 2012). In addition, wastewater pollution can lead to increased concentrations of heavy metals, microplastics, and endocrine disruptors in marsh animals, such as mussels, clams, insects and minnows, which are then likely to bioaccumulate in the food web (Burgos and Rainbow, 2001; Fan *et al.*, 2002; Manzetti and van der Spoel, 2015; Peralta-Videa *et al.*, 2009). Importantly, this increase in metals and toxins has been shown to be detrimental to the performance of both metabolic function and reproduction in animals (Sharma and Agrawal, 2005; Sun *et al.*, 2022). For example, in response to wastewater pollution, mussels and insects produce excess endocrine disruptors in their tissues (Goksøyr, 2006; Petrovic *et al.*, 2002; Weber *et al.*, 2013), which have been associated with reduced reproductive success and deformed reproductive organs. It has also been shown that heavy metal exposure can induce mitochondrial dysfunction (Sun *et al.*, 2022) and inhibit crucial metabolic processes. More recently, it has been observed that wastewater pollution in marsh creeks can lead to blooms of macro- and microalgae which then harm marsh structure and function in two ways: (1) excess algae exhausts all or almost all the available oxygen at night, leading to hypoxia in marsh creeks and decreased health and reproduction of sympatric animals (Cheek *et al*., 2009); and (2) excess macroalgae that is transported onto the marsh at high tides, smothers plants, subsequently causing plant loss on marsh creek banks leading to elevated erosion rates (Wasson *et al*., 2017).

6.13.7 Wastewater Impacts on Bivalve Reef Ecosystems

Across temperate areas of the world, the dominant reef-forming bivalves are oysters and mussels, which generate extensive reefs on the muddy bottoms of estuaries. Mussel and oyster reefs provide important ecosystem functions and services in estuarine environments, including: (1) provisioning of essential habitat for fish (Tolley and Volety, 2005); (2) reducing coastal erosion (Chowdhury *et al*., 2019); (3) increasing estuarine water quality (Cerco and Noel, 2007); (4) producing protein for humans and estuarine food webs (Oakley *et al*., 2014); (5) provisioning of biodiversity through habitat generation (Coen *et al*., 1999); and, (6) acting as hotspots for denitrification (Piehler and Smyth, 2011). Moreover, bivalves are among the most efficient water pumps in nature; for example, one oyster can filter up to 189 liters of water each day, and at their peak biomass (more than 300 years ago) oysters were estimated to have been capable of filtering all the water in the Chesapeake Bay every 3–4 days (Adolf *et al*., 2006).

Despite the wide variety of benefits that bivalve reefs generate, oyster and mussel reefs have experienced the highest decline among all marine habitats due to human impacts (Beck *et al*., 2011). Foremost among those human impacts is overfishing, which has led to an estimated 85% reduction in oyster biomass worldwide (Beck *et al*., 2011). Although overharvesting has been abated in many areas, oyster and mussel reefs have not recovered to anywhere near their former biomass. Many factors are thought to prevent bivalve reemergence, including introduced microbial pathogens, increased predation, and drought-enhanced parasitism (Lenihan, 1999; McCall, 2021). More recently, the role of wastewater pollution has begun to be discussed as a potentially overlooked but important factor that could be preventing bivalve reef regrowth in many areas of the world (Tuholske *et al*., 2021; Wear *et al*., 2021).

6.13.7.1 Human Pathogens and Heavy Metals in Oyster Tissues

The susceptibility of bivalves to wastewater pollutants and associated pathogens has long been the focus of scientific research because bivalves are a food source for many humans, and contamination of their bodies can lead to severe illness and even death in people who consume them (CDC, 1998). Among the most common human pathogens in wastewater are noroviruses and hepatitis A, which can both cause serious gastro-intestinal distress in humans (CDC, 1998). Mathematical models and field correlations show that there is a strong positive relationship between the degree of norovirus contamination in estuarine waters and levels of contamination in the bivalves living in those waters (Razafimahefa *et al*., 2020; Suffredini *et al*., 2014). Furthermore, not only can wastewater pollution contaminate oysters and mussels on bivalve reefs with human pathogens, it can also increase heavy metal concentrations in their tissues (Wang *et al*., 2018). As is the case with human pathogens, the amount of heavy metals found in bivalves is positively associated with the amount of heavy metals found in the wastewater polluting their surrounding aquatic habitat. Myriad studies have shown that oysters near sewage outfalls experience elevated levels of heavy metals in their tissue, including lead, cadmium, zinc, copper, and nickel (Mok *et al*., 2015).

Wastewater pollution can increase the concentration of many other types of contaminants within bivalves. For example, recent studies have shown that while microplastics are found in 94% of oysters around the world, they occur in much higher concentrations in oysters closer to sewage outfalls. These recent studies have also shown that oysters near wastewater pollution sites have higher concentrations of CECs, including alkylbenzene, PFAS, phthalate esters (PAEs), pharmaceuticals, caffeine, pesticides, PCBs, polybrominated diphenyl ethers (PBDEs), PAHs, and flame retardants (Burket *et al*., 2018; Lemos *et al*., 2022). Taken together, these studies show that oysters and mussels are biotic hotspots for concentrating a variety of wastewater contaminants.

6.13.7.2 Impacts on Oyster Health

While most studies of wastewater pollution effects on bivalve reefs have focused on how pollution influences concentrations of contaminants in bivalve tissue, much less is known about how these contaminants affect oyster and mussel health or reef health and growth. Investigations that have studied these response variables in bivalves have shown that exposure to contaminants in wastewater pollution can induce gene expression, skew sex ratios, induce metabolic stress, lower tolerances to low oxygen levels, slow growth, and even lead to death (Blaise *et al*., 2003; Sorini *et al*., 2021). In fact, a study found that even relatively low levels of heavy metal pollution can reduce thermal tolerance in oysters (Lannig *et al*., 2006). The mechanism behind this interaction involves both cadmium and temperature independently decreasing the efficiency of metabolic processes in the oysters' mitochondria. In combination, heavy metals and temperature have a synergistic effect, and oysters exposed to both stressors experience disproportionate increases in disease prevalence and death. Since the primary source of metal contamination in bivalves is wastewater pollution, these findings strongly indicate that the ability of oyster and mussel reefs to resist and recover from large-scale heating and low oxygen events, both of which have been increasing with climate change, could be at risk due to widespread heavy metal contamination in coastal systems from wastewater pollution (Wear *et al*., 2021).

At a larger scale, one of the major threats that wastewater pollution creates for bivalve reefs is low-oxygen waters (Biancani, 2010; Lenihan and Peterson, 1998). In estuarine waters with heavy wastewater pollution, the elevated nutrients and organics in sewage runoff spur massive increases in bacteria, which in turn uptake all, or almost all, the available oxygen in the water as they decompose excess organic matter. For immobile organisms like oysters and mussels, this lack of oxygen can lead to death (Altieri and Witman, 2006). Oyster spat can resettle in these affected areas once low-oxygen stress has abated, but if wastewater pollution persists, then frequent return of low-oxygen events will prevent bivalve reefs from re-establishing. As wastewater pollution is often greater in nearshore estuarine waters compared to offshore environments, sustained pollution-associated stress in these nearshore areas may also lead to range constriction on oysters and greatly limit potential sites available for oyster restoration (Tice-Lewis *et al.*, 2022). For example, the billion-oyster project in New York Harbor, which has a goal of restoring one billion oysters, has been experiencing massive die-offs of restored oysters due, in part, to low-oxygen events induced from wastewater pollution outflows (Baumann *et al.*, 2019). This result is not surprising given that more than 102 billion liters of raw sewage and polluted stormwater discharge flow into New York Harbor each year. Clearly, wastewater pollution suppresses not only individual oyster and mussel health, but also the persistence, growth, and reestablishment of the biogenic reef ecosystems they generate. It is essential that bivalve researchers and managers recalibrate their thinking about oyster and mussel reef ecology and conservation and elevate wastewater pollution to be considered as a major threat to bivalve reefs.

6.13.8 Wastewater Impacts on Finfish

Wastewater pollution contains a cocktail of contaminants that can affect fish across multiple stages of organization, from the cellular level to the ecosystem level. The wide array of pollutants found in wastewater means that fish are exposed to a range of pollutants simultaneously, rendering it challenging to pinpoint the ultimate cause of the observed impacts. As a result, there have been several laboratory studies that have endeavored to isolate pollutant-specific impacts and concentrations at which different responses are observed. Some pollutants have been extensively reviewed and will not be covered here; for instance, a recent review summarized the main effects of suspended solids on fish, finding that sediment can impact the behavior and physiology of fish, and lead to sub-lethal and lethal impacts at high enough concentrations (Wenger *et al.*, 2017). Here, we highlight a few key pollutants (i.e., microplastics, pharmaceuticals, and nutrients) and provide an overview of the effects of those pollutants that have been observed in laboratory and field studies.

6.13.8.1 Microplastics

A recent meta-analysis on the effects of microplastics on fish determined that larval and juvenile fish consumption and feeding were significantly negatively affected by exposure to microplastics (Foley *et al.*, 2018). Indeed, laboratory studies have demonstrated that when small enough, microplastics are more frequently ingested by fish, leading to a negative effect on the physiology, growth, and body condition of the fish (Critchell and Hoogenboom, 2018; Rochman *et al.*, 2014). Once ingested, plastics can remain in the digestive tracts of fish for periods of days to weeks before excretion, and may block digestive tracts or impair digestive function during this time (Foley *et al.*, 2018). Although much of our understanding of the impacts of microplastics on fish comes from laboratory studies, the accumulation of plastic by fish in the wild has been observed. For instance, Rochman *et al.* (2015) found that 28% of fish at markets in Indonesia contained plastic, demonstrating that the trends observed in situ are consistent with responses observed in the laboratory.

Microplastics not only impact fish by disrupting feeding, but they can also accumulate pathogens and contaminants from the water by factors of up to one million times (Wardrop *et al.*, 2016). Previous studies have demonstrated that pollutants concentrated on plastic consumed by fish can lead to significant sublethal impacts (Rochman *et al.*, 2014) and can bioaccumulate in fish tissues (Wardrop *et al.*, 2016), which could have public health implications for consumers of contaminated fish. While not all plastic in the environment is from wastewater pollution, the aforementioned findings demonstrate the potential for wastewater to impair fish in contaminated coastal and marine environments.

6.13.8.2 Pharmaceutical Products

Although pharmaceuticals are designed to target specific chemical pathways in humans, one study found that between 65% and 86% of human drug targets are evolutionarily conserved in 12 diverse fish species, highlighting the possibility of significant impacts on fish (Brown *et al.*, 2014). Consequently, a substantial amount of research has focused on understanding the influence of these substances.

One of the biggest concerns regarding pharmaceuticals in the marine environment is the potential for endocrine disruption. Corcoran *et al.* (2010) reviewed the evidence for ill-health effects on fish from pharmaceutical exposure and found a wide range of physiological impacts across different species and drug classes. For instance, synthetic estrogen at environmentally relevant concentrations induced feminization in multiple fish species. Alarmingly, but not surprisingly, this outcome can result in major population-level impacts. Kidd *et al.* (2007) found that long-term, low-level exposure of the freshwater fathead minnow (*Pimephales promelas*) to synthetic estrogen in an experimental lake caused a complete failure of the fishery. Adding to the concern about pharmaceuticals in the environment, synthetic estrogen can bioaccumulate and concentrate at very high levels, which may in turn impact consumers of contaminated fish (Corcoran *et al.*, 2010).

Although synthetic estrogen seems like the most obvious candidate of pharmaceuticals to cause endocrine disruption, other drugs have also led to endocrine disruption and impacts on reproduction (Corcoran *et al.*, 2010). For instance: non-steroidal anti-inflammatories can disrupt oocyte maturation and ovulation (Lister and van der Kraak, 2008); antidepressants can delay the onset of sexual maturation and disrupt spermatogenesis in male fishes, while reducing egg production in females (Thompson and Vijayan, 2022); and, cholesterol medications can reduce testosterone levels and sperm count (Laville *et al.*, 2004; Runnalls *et al.*, 2007). Pharmaceuticals can also have sublethal and lethal impacts on fish beyond endocrine disruption and impacts on reproduction (Corcoran *et al.*, 2010; Thompson and Vijayan, 2022). Emerging evidence indicates that early life-history stages are most vulnerable to pharmaceuticals (Thompson and Vijayan, 2022), which has also been observed for other pollutants (Wenger *et al.*, 2017). This finding can help practitioners develop targeted management recommendations for wastewater management to protect spawning aggregations and nurseries of commercially and ecologically important fishes.

Traditionally, in routine ecotoxicology studies, direct effects of pollutants on fish physiology and mortality were tested as the primary endpoints, but recent work has recognized the importance of assessing behavioral endpoints as well (Jacquin *et al.*, 2020). Behavioral changes in fish can happen at much lower concentrations than sub-lethal and lethal impacts, and can ultimately lead to population-level changes. Brodin *et al.* (2014) published a comprehensive review on observed behavioral changes induced through exposure to various pharmaceuticals. They report that pharmaceuticals can lead to reduced territorial aggression, feeding rates, and antipredator behavior while also increasing social behavior and boldness (Brodin *et al.*, 2014; references therein). However, the results reported in the studies reviewed by Brodin *et al.* (2014) often vary among species and exposure concentrations, which makes it difficult to generalize overall impacts on fish or extrapolate laboratory studies to in situ conditions. There have been several recent systematic reviews and meta-analyses that have modeled likely exposure thresholds of pollution that elicit a response in different organisms (Nalley *et al.*, 2021; Tuttle and Donahue, 2022; Wenger *et al.*, 2018), and this approach could help to refine our understanding of risk associated with pharmaceutical pollution from wastewater.

6.13.8.3 Nutrients

Most of the research related to nutrient enrichment of coastal environments and subsequent impacts on fish has primarily focused on the impacts of hypoxia caused by nutrient enrichment (Breitburg, 2002). However, nutrients, in and of themselves, have been observed to directly affect fish negatively. Shingles *et al.* (2001) found that when adult trout were exposed to ammonia, they experienced a significant reduction in critical swimming speed and aerobic scope. A similar finding was observed by Tudorache *et al.* (2008), who found that exposure to ammonia reduced both escape performance and predation performance, thus altering predator-prey interactions. Considering that the two primary strategies that fish use in hypoxic conditions are to swim away to find better conditions and to increase ventilation rates (Breitburg, 2002), the direct effects of ammonia on aerobic capacity and swimming performance will consequently compromise these adaptive strategies.

6.13.8.4 *In situ* Observations

Although in-lab toxicity tests are important for understanding how individual pollutants impact fish, they often cannot be extrapolated to field toxicity because of fluctuations in concentrations and interactions with other stressors. Laboratory results can be conclusively linked to a specific pollutant found in wastewater, but in situ studies allow for observation of several types of pollutants and likely multiple sources, and thus, a better understanding of wastewater impacts in the wild to better resolve ecosystem-level impacts.

There are now several studies that have observed impacts on fish in coastal and marine environments that have been linked either to wastewater specifically, or to urbanized environments. One of the most interesting and counterintuitive observations is that fish are often found in greater abundance near outfalls (Azzurro *et al.*, 2010; Grigg, 1994; Guidetti *et al.*, 2003; McCallum *et al.*, 2019; Nikel *et al.*, 2021; Russo, 1982, 1989), potentially due to the presence of a physical structure. For instance, both Russo (1982, 1989) and Grigg (1994) suggest that the new deepwater ocean outfalls create habitats not normally found in those deepwater environments. Others have observed that abundance estimates were driven by more opportunistic species, especially planktivores and particulate organic matter feeders, who were taking advantage of the new food source (Azzurro *et al.*, 2010; Guidetti *et al.*, 2003; Nikel *et al.*, 2021). In fact, a systematic review and meta-analysis found that the average change in fish abundance at wastewater-polluted sites was + 40% (McKinley and Johnston, 2010).

Regardless of the mechanism that is attracting fish to wastewater-polluted areas, there is concern that these sites could act like ecological traps, by congregating fish near outfalls or other contaminated locations and subsequently exposing them to increased levels of harmful pollutants (Gray, 1996; McCallum *et al.*, 2019; Nikel *et al.*, 2021). For instance, Corbett *et al.* (2015) found a greater severity of physiological impacts on fish near a wastewater outfall, which decreased with distance. Additionally, Schlacher *et al.* (2007) found similar impacts on fish from a sewage-contaminated estuary in comparison to those from a clean reference site. In another study, fish from a large outfall location were significantly smaller and had smaller eggs in comparison to fish from a control location (Smith and Suthers, 1999).

The findings of sub-lethal impacts on fish exposed to wastewater pollution indicate why some studies have reported reduced fish abundance and species richness at outfalls (Reopanichkul *et al.*, 2009; Smith *et al.*, 1999). It is likely that differences in the type of treatment, the volume of the effluent runoff, and the types of pollutants discharged, all influence the patterns observed. If the discharged materials or the concentration of pollutants exceed a certain threshold, species abundance may begin to decrease due to the progressive disappearance of pollution-sensitive species. More research needs to be conducted on patterns of fish distribution and health in relation to varying wastewater pollution levels to better assess this risk.

6.13.8.5 Fish Kills

Large-scale fish mortality events (i.e., fish kills) have been observed globally. In many cases, there are multiple contributing factors that precipitate the event, making it hard to pinpoint one specific cause. Still, there are now several fish kill events from around the world that have been more conclusively linked to wastewater pollution. In many cases, the fish kills were related to either hypoxic events associated with high nutrient concentrations from untreated wastewater (e.g., the multiple fish kill events observed in the Tapi Estuary in India [Ram *et al.*, 2014]), or dinoflagellate blooms from nutrient enrichment (e.g., Kuwait Bay in 1999 [Heil *et al.*, 2001]). In other cases, wastewater was a key contributing factor among others. A study of the sources and causes of fish kills in Texas, U.S.A. from 1951 to 2006, found that the majority of fish kill events in bays (57%) were caused by low dissolved oxygen concentrations. The hypoxic conditions primarily occurred in water bodies constructed for residential or industrial purposes where local circumstances, including hot weather and seepage from residential septic systems near the canals, caused the dissolved oxygen levels to plummet, killing schools of fish that were unable to move out of the area quickly enough (Thronson and Quigg, 2008).

Wastewater can also introduce pathogens into the marine environment or promote the growth, dissemination, and virulence of pathogens. A forensic assessment following a massive fish kill event in Kuwait Bay in 2001 found that a bacterial pathogen isolated from an infected fish matched samples collected from wastewater outfalls, implicating sewage as the source of the contamination (Al-Marzouk *et al.*, 2005). The introduction of novel or antibiotic-resistant bacteria has the potential to create significant problems for coastal fisheries; in fact, antibiotic-resistant bacteria have already been observed in commercial fisheries in Chile near wastewater discharge outfalls (Miranda and Zemelman, 2001). In addition, the elevated nutrient and pollutant levels in wastewater can also weaken the immune systems of fish, rendering them more vulnerable to pathogens (Corcoran *et al.*, 2010; Glibert *et al.*, 2002). In Egypt, wastewater pollution and unfavorable environmental conditions were implicated in a massive fish kill event by triggering bacterial infections (Eissa *et al.*, 2021) and also possibly suppressing the immune system of fish, thus making them more vulnerable to infection. The massive fish kill events described above highlight the economic consequences of poor wastewater management. Not only have these events caused massive mortality in commercially and recreationally important species (Eissa *et al.*, 2021; Al-Marzouk *et al.*, 2005; Thronson and Quigg, 2008), but they have also resulted in the closure of fisheries to avoid public health risks (Al-Yamani *et al.*, 2020; Heil *et al.*, 2001).

6.13.9 Wastewater Impacts on Marine Mammals

Marine mammals form a diverse community of both fully aquatic and amphibious mammal species that inhabit, and depend on, both coastal and offshore regions. While some species are exclusively pelagic or coastal, the majority of species (especially cetaceans and pinnipeds) utilize both habitats (Würsig *et al.*, 2009). While the global scale of wastewater pollution in marine environments indicates that most marine mammal species are at some risk of exposure, the susceptibility to particular pollutants is dependent on the species' ecology (Desforges *et al.*, 2016; Tuholske *et al.*, 2021; Wear *et al.*, 2021). Marine mammals have a wide breadth of behaviors, life histories, foraging strategies, habitats, and physiologies that influence how they interact and are affected by pollutants introduced to the environment.

6.13.9.1 Nutrients and Harmful Algal Blooms

Wastewater-borne pollutants can affect marine mammals both indirectly and directly. Indirect effects include ecosystem disruption, or disturbance to environmental facets that marine mammals depend on, such as prey availability and quality. In coastal environments, the outflow of excess nutrients (e.g., nitrogen and phosphorus) can result in eutrophication, and subsequently, harmful algal blooms. For example, diatoms of the genus *Pseudo-nitzschia*, whose abundances are related to nutrient fluxes (Parsons and Dortch, 2002; Van Meerssche *et al.*, 2018), produce a toxin, domoic acid, which commonly impacts California sea lionizers (*Zalophus californianus*) causing severe neurological symptoms (Goldstein *et al.*, 2008). California sea lionizers with domoic acid toxicosis experience acute and chronic effects including hippocampal atrophy (Goldstein *et al.*, 2008; Cook *et al.*, 2016), degenerative cardiomyopathy (Zabka *et al.*, 2009), and the presence of domoic acid within milk of lactating mothers, resulting in vertical transmission of the toxin (Rust *et al.*, 2014). While California sea lionizers have the most extensive reporting regarding domoic acid, the toxin has been found in a suite of other marine mammal species, such as northern fur seals (*Callorhinus ursinus*) (Lefebvre and Robertson, 2010), harbor seals (*Phoca vitulina*) (McHuron *et al.*, 2013), southern sea otters (*Enhydra lutris nereis*) (Kreuder *et al.*, 2003), humpback whales (*Megaptera novaeangliae*), and blue whales (*Balaenoptera musculus*) (Lefebvre *et al.*, 2002). While algal blooms represent an example of an acute outcome, wastewater effluent can also cause long-term impacts on marine mammal habitats. The endangered west African manatee (*Trichechus senegalensis*), which depends on macrophytes as a food source, is a population at risk, in part due to a reduction in macrophyte habitat impacted by anthropogenic nutrient input (Takoukam *et al.*, 2021). While most studies do not address causation between wastewater effluent and its effects on marine mammals, the literature clearly demonstrates that wastewater input is a contributor to nutrient flux in coastal ecosystems (Tuholske *et al.*, 2021; Von Sperling, 2007; Wear *et al.*, 2021), potentially impacting population health and reproductive capacity of free-living populations of marine mammals and the habitats they depend on.

6.13.9.2 Pollutants and Pathogens

Direct effects of wastewater pollutants on marine mammals involve the consumption of pollutant-laden prey, drinking contaminated seawater, and interactions between exposed body parts and pollutants in the environment. Over time, repeated ingestion of pollutants accumulates within various tissues of the body, increasing in concentration through bioaccumulation. Diet composition is a key determinant of pollutant concentrations in marine mammal tissues; however, diets vary greatly among marine mammals, and thus, so do their exposure to pollutants (Reijnders *et al.*, 2009). Diets can also vary within a species, such as in the northern elephant seal (*Mirounga angustirostrus*), where prey type and foraging behavior are dependent on sex and demographic class (Field *et al.*, 2005; Kienle *et al.*, 2022; Le Boeuf *et al.*, 2000). Marine mammals that forage on higher trophic level prey, which contain high concentrations of pollutants due to biomagnification (Gray, 2002), are especially at risk of developing high pollutant loads within their tissues. Persistent organic pollutants (POPs) that can be sourced from wastewater are of particular concern to marine mammals; these include but are not limited to organotin compounds, heavy metals, organochlorine pesticides, PFAS, PCBs, PBDEs, and microplastics (Simmonds, 2017; Würsig *et al.*, 2009). In marine mammals, POPs act as immunosuppressors and hormone imitators, resulting in susceptibility to disease, sensitivity to physiological stress, and abnormalities in reproduction (Jepson and Law, 2016; Parsons *et al.*, 2013; Reijnders, 1996; Sonne *et al.*, 2020). In the United Kingdom, necropsies performed on stranded harbor porpoises (*Phocoena phocoena*) found PCB load to be a significant predictor of reproductive status, in which almost 20% of the total 329 porpoises examined had reproductive failure (Murphy *et al.*, 2015). Similar results were also found in common dolphins (*Delphinus delphis*), where over 16% of sampled females exhibited reproductive failure, of which higher blubber PCB concentration was a significant predictor (Murphy *et al.*, 2018). Wastewater POP input into the coastal environment represents an ongoing and global threat to marine mammal health; however, POPs represent only a subset of the potential direct effects of wastewater pollution.

Wastewater pollution introduces zoonotic pathogens into the coastal marine environment, risking transmission and infection to marine mammals that can cause physiological stress and other deleterious health effects. Wastewater effluent commonly includes helminth parasites and bacterial, viral, and protozoan pathogens that are also found in free-living marine mammals (Adell *et al.*, 2016; Fayer *et al.*, 2004; Von Sperling, 2007). Examples include *Escherichia coli* infections in dolphin species (Li *et al.*, 2021), *Giardia duodenalis* infections in pinnipeds (Ebmer *et al.*, 2020; Lehnert *et al.*, 2019), and toxoplasmosis (*Toxoplasma gondii*) infections in southern sea otters (Adell *et al.*, 2016; Miller *et al.*, 2008). Wastewater-borne pathogens contribute an additive stressor that can negatively impact the health and reproduction of an affected population, and in some cases cause individual fatalities within vulnerable species, as has been reported with toxoplasmosis infections in endangered southern sea otters and Hawaiian monk seals (*Neomonachus schauinslandi*) (Dubey *et al.*, 2020). While information regarding transmission pathways between wastewater and marine mammals is currently lacking or contradictory, evidence is clear that pathogens detrimentally impact the health of marine mammals (Adell *et al.*, 2016; Forman *et al.*, 2009; Miller *et al.*, 2008; Schaefer *et al.*, 2011) and that wastewater is a global source (Payment *et al.*, 2001; Tuholske *et al.*, 2021).

Marine mammals are an animal group with diverse life-history strategies, distribution and movement patterns, diet, ecology, and susceptibility to marine pollution. While it is difficult to attribute a specific physiological effect to a single pollutant or source in free-living animals, recent work using meta-analyses of case studies and collections of work on identified 'sentinel species' presents a more robust understanding of the health of marine mammals to specific pollutants that can be emitted from wastewater (Dubey *et al.*, 2020; Fossi and Panti, 2017; Seguel and Gottdenker, 2017; Sonne *et al.*, 2020). The nature in which pollutants interact with marine mammal physiology is complex and must be studied and viewed in a holistic sense, by acknowledging that accumulating pollutants are additive stressors to the health of an individual animal. Stressors introduced by wastewater pollution can render recovery from natural stressors more difficult for an individual, despite the pollutants being benign in isolation. Additionally, how wastewater impacts will change with climate change is virtually unknown in marine mammals. Changes in pollutant loads, concentrations, and treatment rates in the near future could dramatically influence wastewater pollution effects on coastal ecosystems and communities (Jenssen, 2006). Wastewater pollution, while ubiquitous in almost all coastal systems, is severely understudied (especially as related to marine mammals) and deserves additional attention in future research.

6.13.10 Solutions

The development of wastewater treatment infrastructure and discharge regulations has historically focused on public health alone — a legacy that has lasted well into the twenty-first century (De Feo *et al.*, 2014; Lofrano and Brown, 2010). However, emphasis on the removal of human pathogens has obfuscated the fact that sewer systems are amassing and concentrating many other contaminants (Blair *et al.*, 2017, Liu *et al.*, 2019b; Tuholske *et al.*, 2021; Van Puijenbroek *et al.*, 2019) and releasing them directly into the environment. Wastewater treatment processes are now under greater scrutiny by traditionally unaffiliated disciplines in recognition of wastewater's connections to ecological health, economic stability, resource security, social equity issues, and more.

In light of growing awareness and new research unveiling the impacts of wastewater pollution, a solution space has begun to emerge beyond the siloed approach of ecosystem-based management strategies and the traditional avenues of policy, governance, and administrative controls. Voltaire's quote, "Perfect is the enemy of the good," serves as a necessary reminder that implementing some kind of sanitation infrastructure is better than having none at all. No "one size fits all" solution exists; instead, assessing localized factors (e.g., geology, climate, and available space), as well as understanding the socioeconomic context (e.g., cultural norms and cost analysis) can help inform the decision-making process and determine optimal trade-offs for a given circumstance. Sanitation planning requires a strategic, community-based approach to allow for successful implementation and system adoption.

The ability to treat CECs is possible via tertiary treatment techniques like reduced osmosis, carbon filtration, advanced ozonation, and nanofiltration, among others. For example, one on-site treatment design, nitrogen removing biofilters, uses soil microbes to remove up to 90% of nitrogen and organic contaminants (Gobler *et al.*, 2021), as well as novel contaminants including pharmaceuticals, personal care products, and the carcinogen 1,4-dioxane (Clyde *et al.*, 2021; Gobler *et al.*, 2021; Lee *et al.*, 2021). High costs may impede the adoption of some advanced technologies, but general awareness and policy may fuel action, as concerns around "forever chemicals" and PFAS grow in the public sphere. Additionally, it has been suggested that water reuse will drive the adoption of advanced treatment technologies, especially in circumstances where wastewater contamination threatens an area's water supply (RTI Innovation Advisors, 2021). Water reclamation and other forms of resource recovery, including urine diversion for renewable fertilizer and conversion of solid waste into biochar and biofuel, offer opportunities to divert wastewater effluent and utilize wastewater components before they reach the environment.

Gray infrastructure is not the only solution; it is possible to harness natural ecological processes for contaminant removal and to integrate green infrastructure for greater ecological resilience. As mentioned earlier, nutrient-limited ecosystems (e.g., seagrasses, salt marshes, and mangroves) may not withstand wastewater effluent on their own without experiencing repercussions, but their natural functions can aid in the wastewater treatment process. Constructed wetlands, also known as treatment wetlands, are designed to integrate the biological, physical, and chemical interactions that naturally occur in wetland ecosystems. They serve as effective biofiltration systems for excess sedimentation, nutrients, and organic matter (Yang *et al.*, 2008), and even have the ability to effectively remove novel contaminants (Ávila *et al.*, 2015; Matamoros *et al.*, 2017). Similarly, bivalve filtering, although unsustainable when used beyond bivalves' biological limits, can serve as a biological purification process to be used in tandem with wastewater treatment technologies (Gudimov, 2021). Seagrass plants also show promising potential for mitigating a range of constituents found in wastewater pollution. Products derived from seagrass plants can be used during wastewater treatment to remove potentially harmful compounds. For example, Neptune grass (*Posidonia oceanica*) can adsorb the antibiotic oxytetracycline (Ferchichi *et al.*, 2022); activated coastal waste from eelgrass (*Zostera marina*) can remove manganese compounds (Deniz and Erslani, 2020); and, little Neptune grass (*Cymodocea nodosa*) can remove cadmium and nickel ions from solution (Moawad *et al.*, 2020). The use of seagrass-derived products could make wastewater treatment processes and costs more efficient and effective, and less environmentally damaging. Seagrass plants and ecosystems have also been shown to filter pathogens and pollutants in wastewater that impact human health. For example, seagrass leaves and their associated epiphytes can remove pollutants such as microplastics from the water column (Goss *et al.*, 2018, Sanchez-Vidal *et al.*, 2021). Additionally, as pathogen filtration systems, seagrass ecosystems reduce levels of potentially pathogenic bacteria in the water column by as much as 50% (Lamb *et al.*, 2017; Palazón *et al.*, 2017; Reusch *et al.*, 2021). Seagrass ecosystems are conservatively estimated to save USD $24 million each year in illness-associated costs and prevent 8 million cases of illness annually (Ascioti *et al.*, 2022).

Nature-based solutions are potential pathways for updating antiquated sanitation systems, sometimes with the benefit of simpler maintenance requirements and reduced costs (Risch *et al.*, 2021). Studies have indicated that those solutions offer other co-benefits, including greater biodiversity, pollination, carbon sequestration, temperature regulation, biomass and biosolid production, water availability, and food production (Cross *et al.*, 2021). In addition to providing esthetic value and recreational sites, living shorelines and constructed wetlands can enhance coastal resilience, by mitigating floods and storm surge (services that have the potential to assist with climate adaptation efforts for strategically selected, small geographic areas). Algal turf scrubbers, algal bioreactors, and microalgae biotechnology have great potential as treatment alternatives for reducing energy consumption, capturing carbon dioxide, and recovering organic nutrients for reuse, despite some challenges to large-scale implementation (e.g., land requirements and operations). However, the decision-making process of selecting an appropriate solution is fundamentally circumstance-specific and requires weighing of pros and cons; for instance, living shorelines cannot be used in areas with strong currents and heavy wave activity. Budgets for construction vary based on available labor and expertise, as well as land and material availability. Another major concern is ensuring that a project has the capacity and capital for operations and maintenance beyond the upfront costs; to return to the living shoreline example, marsh platforms require extensive upkeep, as they need repeated thin-layer applications of sediment. With all these considerations in mind, there is no single path forward to safe, sustainable sanitation: sometimes gray infrastructure is the most pragmatic option, but other times, green and hybrid approaches serve as viable, potentially advantageous, alternatives.

6.13.11 Conclusions

This cross-ecosystem synthesis reveals new insights into the extent and details of how wastewater pollution impacts marine and estuarine ecosystems. Most importantly, our review highlights that: (1) wastewater pollution is much more extensive than currently realized (e.g., it reaches over 30% of salt marshes globally [Wear *et al.*, 2021]); and, (2) wastewater pollution impacts on ecosystems are generated not only by excess nutrients but also by a great assortment of other contaminants that are found in wastewater. Nutrient enrichment of coastal ecosystems generates a range of harmful impacts, including overgrowth of corals by algae, nitrate toxicity in corals, smothering of marsh plants by floating algal mats, increased edge erosion in salt marshes and mangroves, increased vulnerability of mangroves to climatic stressors, and death of fish and invertebrates due to nutrient-fueled anoxic events in estuarine waters. Eutrophication is one of the most serious impacts in estuaries worldwide. The non-nutrient pollutants found in sewage, such as heavy metals, human pathogens, pharmaceuticals, and endocrine disruptors, are leading to feminization in estuarine fishes, increased vulnerability to heat stress, direct death of oysters and corals, suppressed reproductive success in invertebrates and fish, and increased disease in fish, corals, and other animals. These effects can interact with other human and climate change impacts, such as overfishing and increasing temperature, to lead to synergistic effects on ecosystem function. For example, wastewater pollution and warming waters can synergistically increase the incidence of harmful algal blooms, while wastewater pollution and overharvesting of top predators can lead to synergistic increases in edge erosion in salt marshes and leave them more vulnerable to sea level rise. Not only does this new understanding challenge current thought that wastewater impacts are limited, but it also overturns long-held beliefs in salt marsh and mangrove ecology that these wetland systems benefit from wastewater effluent because they are nutrient-limited. Indeed, we now know that salt marshes and mangroves that experience wastewater pollution are likely to decrease in elevation, experience higher creek bank erosion, accumulate heavy metals in animals that are often eaten by humans, and become more vulnerable to drought and storms. It is paramount that scientists and managers recalibrate to this new understanding that wastewater pollution is vast in spatial extent and strong in impact and to also elevate its status as a major threat to marine and estuarine habitats and organisms. Solving the wastewater pollution threat is not simple, but solutions are available, and reducing wastewater impacts will increase coastal ecosystem resilience to global change.

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Further Reading

A practitioner's guide for ocean wastewater pollution, https://drive.google.com/file/d/1g0LCWLvsN182sh-JQCynnK8RKqXGIG93/view?usp=sharing.

Relevant Websites

https://www.oceansewagealliance.org
Ocean Sewage Alliance.