COASTAL FORUM:

Human and ecosystem health in coastal systems

By

Nicole Elko,[1] Diane Foster,[2] Gregory Kleinheinz,[3] Britt Raubenheimer,[4] Susanne Brander,[5] Julie Kinzelman,[6] Jacob P. Kritzer,[7] Daphne Munroe,[8] Curt Storlazzi,[9] Martha Sutula,[10] Annie Mercer,[11] Scott Coffin,[12] Carolyn Fraioli,[13] Luke Ginger,[14] Elise Morrison,[15] Gabrielle Parent-Doliner,[16] Cigdem Akan,[17] Alberto Canestrelli,[18] Michelle DiBenedetto,[19] Jackelyn Lang,[20] and Jonathan Simm[21]

1) nicole.elko@asbpa.org, American Shore and Beach Preservation Association, P.O. Box 1451, Folly Beach, SC 29439
2) University of New Hampshire, School of Marine Science and Ocean Engineering, 24 Colovos Drive, Durham NH 03824
3) University of Wisconsin-Oshkosh, 800 Algoma Boulevard, Oshkosh, WI 54901
4) Woods Hole Oceanographic Institution, MS11 266 Woods Hole Road, Woods Hole, MA 02543
5) Oregon State University, 2030 SE Marine Science Drive, Newport, OR 97365
6) University of Wisconsin-Parkside, 900 Wood Road, P.O. Box 2000, Kenosha, WI 53141
7) Northeastern Regional Association of Coastal Ocean Observing Systems, 195 New Hampshire Avenue #240, Portsmouth, NH 03801
8) Rutgers University, Haskin Shellfish Research Lab, 6959 Miller Ave, Port Norris, NJ 08349
9) U.S. Geological Survey, Pacific Coastal and Marine Science Center, 2885 Mission Street, Santa Cruz, CA 95060
10) Southern California Coastal Water Research Project Authority, 3535 Harbor Blvd., Suite 110, Costa Mesa CA 92626
11) American Shore and Beach Preservation Association, 1509 George II Hwy SE, Bolivia, NC 28422
12) California State Water Resources Control Board, 1001 I Street, Sacramento, CA 95814
13) New York State Dept. of State, 99 Washington Avenue, Suite 1010, Albany, NY 12231
14) Heal the Bay, 1444 9th Street, Santa Monica, CA 90401
15) University of Florida, Dept. of Environmental Engineering Sciences, 1949 Stadium Road, P.O. Box 116580, Gainesville, FL 32611-6580
16) Water Rangers, 123 Slater Street, 6th Floor, Ottawa, Ontario, K1P 5H2, Canada
17) University of Northern Florida, 1 UNF Drive, Jacksonville, FL 32224
18) University of Florida, Weil Hall 575J, 1949 Stadium Road, P.O. Box 116580, Gainesville, FL 32611-6580
19) University of Washington, 3900 East Stevens Way NE, Seattle, WA 98195
20) University of California Davis, 1275 Med Science Drive, Tupper Hall 2108, Davis, CA 95616
21) H.R. Wallingford, Howbery Park, Wallingford, Oxfordshire OX10 8BA, United Kingdom

ABSTRACT

U.S. coastal economies and communities are facing an unprecedented and growing number of impacts to coastal ecosystems including beach and fishery closures, harmful algal blooms, loss of critical habitat, as well as shoreline damage. This paper synthesizes our present understanding of the dynamics of human and ecosystem health in coastal systems with a focus on the need to better understand nearshore physical process interactions with coastal pollutants and ecosystems (e.g. fate and transport, circulation, depositional environment, climate change). It is organized around two major topical areas and six subtopic areas: 1) Identifying and mitigating coastal pollutants, including fecal pollution, nutrients and harmful algal blooms, and microplastics; and 2) Resilient coastal ecosystems, which focuses on coastal fisheries, shellfish and natural and nature-based features (NNBF). Societal needs and the tools and technologies needed to address them are discussed for each subtopic. Recommendations for scientific research, observations, community engagement, and policies aim to help prioritize future research and investments. A better understanding of coastal physical processes and interactions with coastal pollutants and resilient ecosystems (e.g. fate and transport, circulation, depositional environment, climate change) is a critical need. Other research recommendations include the need to quantify potential threats to human and ecosystem health through accurate risk assessments and to quantify the resulting hazard risk reduction of natural and nature-based features; improve pollutant and ecosystem impacts forecasting by integrating frequent and new data points into existing and novel models; collect environmental data to calibrate and validate models to predict future impacts on coastal ecosystems and their evolution due to anthropogenic stressors (land-based pollution, overfishing, coastal development), climate change, and sea level rise; and develop lower cost and rapid response tools to help coastal managers better respond to pollutant and ecosystem threats.

Worldwide, almost 1 billion people live at elevations within 10 m of present sea level (Kulp and Strauss 2019). Our coastal communities and ecosystems are increasingly threatened by nutrients, pathogens, and other contaminants associated with a range of geophysical and human pressures including, but not limited to, warming temperatures, rising sea/lake levels, increasing frequency of extreme storm events, and expanding coastal populations. These pressures are resulting in an unprecedented and growing number of impacts to coastal ecosystems including beach and fishery closures, harmful algal blooms, loss of critical habitat, as well as shoreline damage further impacting coastal economies and communities. United States (U.S.) governmental agencies (National Institutes of Health [NIH], National Science Foundation [NSF], National Oceanic and Atmospheric Administration [NOAA], Environmental Protection Agency [EPA], U.S. Army Corps of Engineers [USACE], and U.S. Geological Survey [USGS]), recognizing the link between coastal physical processes and human and ecosystem health is of critical importance, prioritize the need to develop synergies in funding coastal research across topical areas.

The nearshore, a transition region between land and the continental shelf including (from onshore to offshore) coastal plains, wetlands, estuaries, coastal cliffs, dunes, beaches, surf zones (regions of wave breaking), and the inner shelf (Elko *et al.* 2015), is under threat from sea level rise, long-term erosion, extreme storms, and anthropogenic influences, which affect water quality and impact ecosystems and human health. The interactions between water, sediment, and biota in nearshore systems present challenges associated with assessing human and ecosystem health and identifying solutions. The system exhibits high spatial and temporal variability associated with changing hydrodynamics (e.g. waves, storm surges, tides, runoff, or sea level rise) and biogeochemical forcing (e.g. warming waters, invasive species, and point source pollutants) forcing. Combined, this variable forcing presents significant challenges associated with measuring and predicting water quality, predicting the fate and transport of pollutants, and building resiliency in our coastal ecosystems and communities.

To advance current understanding of human and ecosystem health in coastal systems and determine future needs, the U.S. Coastal Research Program (USCRP) hosted a virtual workshop series in January 2021. The goal of the workshop and the subsequent synthesis was to offer insight into societal challenges and inspire the next generation of research into solutions associated with ensuring human and ecosystem health. Workshop attendees included academics, federal agency scientists and engineers, non-governmental organizations (NGOs), state and regional agencies, private industry scientists and engineers, and local coastal managers. Following the workshop, attendees were invited to participate in an online poll to collaboratively prioritize societal needs and tools/technologies needed to address them for future research investment. The workshop identified key management challenges and high priority federal agency needs to be addressed by coastal science research at the intersection of coastal physical processes and human and ecosystem health. This paper will examine the issues impacting the human and ecosystem health in coastal systems. It is organized around two major topical areas and six subtopic areas: 1) Identifying and mitigating coastal pollutants, including fecal pollution, nutrients and harmful algal blooms, and microplastics; and 2) Resilient coastal ecosystems, which focuses on coastal fisheries, shellfish and natural and nature-based features (NNBF). It represents a collaborative stakeholder perspective of some of the societal needs and tools/technologies required to address these six subtopic areas within human and ecosystem health in coastal systems. The paper provides summary recommendations for scientific research, infrastructure investments, community engagement, and policies to address coastal management challenges.

IDENTIFYING AND MITIGATING COASTAL POLLUTANTS

Fecal pollution

Lead co-author: Julie Kinzelman

Enteric pathogens, a concern to human, animal, and environmental health, were the predominant cause of untreated recreational water-associated outbreaks reported to the Centers for Disease Control and Prevention (CDC) from 2000-2014 (Graciaa *et al.* 2018). These cases were attributed to transmission via ingestion of contaminated water, likely from multiple fecal sources including swimmers, storm water runoff, sewage overflows, septic systems, animal waste, or boating waste. Once present, bacterial pathogens have been found to persist in ocean waters (Yamahara *et al.* 2007; Goodwin and Pobuda 2009; Halliday and Gast 2011), the Great Lakes, inland freshwater (Wiedenmann *et al.* 2006), and beach sands (Ge *et al.* 2010; Weiskerger *et al.* 2019), likely posing a human health risk (Heaney *et al.* 2012). Gastrointestinal illness from exposure to microbial pathogens in U.S. coastal waters is estimated to cost approximately $3 billion annually (DeFlorio-Baker *et al.* 2018).

The EPA Recreational Water Quality Criteria recommends using fecal indicator bacteria (FIB), *Escherichia coli* (*E. coli*) and enterococci, as measures of fecal pollution in fresh and marine water, respectively (EPA 2012). Monitoring fecal indicator organisms is the first line of defense in the protection of public health; however, these techniques do not provide source attribution or details regarding the transport mechanisms, persistence, and propagation of fecal pollution. Since fecal bacteria can persist in sediments without fecal pollution, they can lead to false positives for FIB counts. Additionally, many fecal microorganisms are not pathogenic, and FIB counts do not capture the pathogenic viruses found in fecal pollution (Symonds *et al.* 2009), they do not necessarily translate into adverse health outcomes and can result in the closures of beaches or shellfisheries that may not be a threat to public health.

Here, we discuss recent advances in tools for monitoring fecal pollution, including viruses and microbial community profiling, and recent EPA-approved methods for identifying sources of fecal pollution. We highlight innovations made in modeling fecal pollution's persistence and transport, and the successful approaches used to communicate water quality measures to the public. These advances have provided new insights into the prevalence and persistence of fecal pollution in recreational waters in the U.S. However, communities need further development in monitoring and modeling approaches, particularly in the context of severe weather events and flooding.

SOCIETAL NEEDS & TOOLS/ TECHNOLOGIES NEEDED TO ADDRESS THEM

Source identification

Identifying and mitigating coastal pollutants, including fecal pollution, requires a robust set of tools not only protective of public health in real or near real time, but also capable of distinguishing or attributing the parent sources. While traditional regulatory monitoring approaches remain relevant and useful, source identification and mitigation are the best and truest form of public health protection because they provide a level of permanency rather than singular avoidance in response to instances of water quality failure.

Assessment of FIB (e.g. fecal coliforms, *E. coli*, and enterococci) have long been the foundation of recreational water quality and shellfish monitoring programs but do not differentiate between sources of pollution. Microbial source tracking (MST) methods build upon traditional FIB assessments by attributing these bacteria to their point of origin (e.g. human sewage, dogs, seagulls). Early iterations of MST employed library-dependent (Mott and Smith 2011) and library-independent methods (Wuertz *et al.* 2011), relying on genotypic (library-dependent) or phenotypic (library-independent) characteristics of bacterial isolates in comparison to known sources. Later technologies employed species specific markers (e.g. human specific HF183) to directly attribute pollution sources, providing an advantage over fecal indicator organisms assessments alone (Harwood *et al.* 2014; Ahmed *et al.* 2019). Human-associated methods for viruses, such as the novel bacteriophage crAssphage, have also been targeted as another human-associated method for environmental water quality testing (Stachler *et al.* 2018; Korajkic *et al.* 2020), and the EPA has recently approved two standardized quantitative polymerase chain reaction (qPCR) methods for the characterization of human fecal pollution in water (Methods 1696 and 1697), which have advanced the standardization of MST methods. However, there are still methodological limitations with some MST approaches. For instance, human associated *Bacteroides* spp. targets (HF183/ BacR287 and HumM2) are not able to differentiate between different sources of human fecal pollution (i.e. septic versus sewer systems), which need to be considered when selecting an MST method. The development of metagenomic methods (e.g. bacterial community profiling and next generation sequencing) provide additional insights into the ecology of microbially mediated processes influencing water quality such as harmful algal blooms, fate and transport of contaminants, and pathogen dissemination (BoonFei *et al.* 2015). The combination of approaches, in conjunction with enhanced monitoring/data analysis, and the standardization of MST methods, brings us closer to addressing the challenge of fecal pollution source identification.

Enhancing observations

Whether conducting traditional FIB assessments or employing MST techniques for the investigation of pollution sources, the time interval from sample collection to results and data reporting has typically been on a scale of days rather than minutes or hours. From a public health perspective, this delay results in monitoring authorities taking retrospective action at recreational beaches, failing to reduce exposure risk, and realizing economic loss due to loss of utility even though water quality may have improved. A lengthy delay in results can also hamper the investigative process. In addition, MST and metagenomic analyses require specialized equipment and highly trained staff, both costly and perhaps out of reach for many end users due to budgetary constraints.

Molecular methods (e.g. qPCR) provide rapid options for sampling/ detection. QPCR, which relies on quantification of DNA rather than growth of microorganisms on selective media, can reduce the laboratory turnaround time from 18-24 hours to as little as three, improving the capability of regulatory authorities to manage risks to recreation and shellfish harvesting (Holcomb and Stewart 2020; Dorevitch *et al.* 2017; Kinzelman *et al.* 2011; Lavender and Kinzelman 2009). However, rapid molecular methods are still more costly than traditional culture-based assays and require specialized equipment and trained staff. The standardization of methods may further promote the widespread adoption of these methods.

Environmental, predictive, nowcast, or forecast models provide rapidity but with lower operational and management costs, once constructed and validated. Predictive models have been used for assessing recreational water quality (Coles and Bush 2019; Francy *et al.* 2013) as well as for MST (Kim *et al.* 2018; Whelan *et al.* 2018). Additionally, MST has been incorporated into quantitative microbial risk assessments (QMRA) to estimate public health risk associated with fecal pollution, although challenges with implementing this approach still remain (Zimmer-Faust *et al.* 2020). Fecal indicator organisms are frequently incorporated into models, in addition to ambient environmental conditions gathered through a sanitary survey process (Morris 2013). Research is needed to improve these models, as is discussed in more detail later in this section.

Fate & transport

Physical coastal processes are important for determining the fate and transport of fecal pollutants. When assessing the fate and transport of fecal pollution, it is important to consider the matrix of interest, since fecal pollution can be partitioned between sand and water. The interaction between these matrices can influence the transport and persistence of fecal pollution in recreational settings, and beach sediments can serve as sources of FIB. Since most FIB are particle-associated, resuspension events driven by waves and storms, as well as runoff inputs, can be important factors for the transport of fecal pollution (Fries *et al.* 2006). In general, FIB in water are indicative of short-term conditions, while FIB in sediment are representative of long-term conditions (Kinzelman *et al.* 2020). Particle dynamics, intermittent or persistent pollution sources, solar insolation, rainfall, salinity, temperature, and wave energy can influence the persistence of fecal pollution in the environment and must be considered as well (Feng *et al.* 2015; 2013).

Improved modeling

Physical processes in estuaries and the coastal region (waves and currents), biological factors, (growth, mortality, biofilms, predation), and the spatial and temporal extent of FIB sources, determine fate and transport of FIB entering coastal waters. Models of these processes can be valuable as rapid decision support tools for fecal pollution in recreational waters.

Models can be usually divided in two categories: process-based and statistical models. Process based models are in turn divided into hydrological models (surface run-off and groundwater) and estuarine/coastal hydrodynamic models. Hydrological models, which are usually designed to work at the watershed scale, need information on land use, soil type, fecal scat presence, as well as land topography, and use simplified equations for the transport of pollutants (Nevers and Boehm 2010). In recent years, high-resolution variably saturated groundwater models have been used to solve for the transport and decay of pollutants released by onsite and decentralized wastewater treatment systems (Dong *et al.* 2019). However, due to the high computational cost, these models are mainly used to explain small scale processes and define parameterizations that can be used by more efficient steady-state, depth-averaged models, which operate at a larger scale (Rios *et al.* 2013). The loads computed by hydrological models are used as boundary conditions for estuarine/coastal process-based models solving for microbial transport and include FIB decay rates and particle dynamics, such as settling, accumulation, transport, and resuspension (Thupaki *et al.* 2013; Huang *et al.* 2017; Nevers *et al.* 2020). Although process-based models are a valuable tool to determine the fate of pollutants and identify the sources most impacting coastal water quality, they: (1) may not include or resolve all of the processes important to the fate of pollutants; (2) may not always provide information at the resolution needed for management decisions; (3) require calibration with FIB concentrations measured at several locations and during different hydrological and hydrodynamic conditions, in order to reduce prediction errors; and (4) are usually too computationally expensive for real-time predictions. Research to improve these models may enable mitigation of some of these issues.

Statistical models are a valuable alternative for real-time predictions. These models are usually designed for a specific location and trained by a sufficiently long (i.e. years/decades) time series of FIB concentrations and environmental predictors (Searcy *et al.* 2018). These predictors can be hydrodynamic (water temperature, wave period, height, and direction, river/stream flow rates) or atmospheric (rainfall, wind speed/direction, pressure, cloud cover, air temperature, dew point). As with the process-based models, improvements to the physical and biological underpinnings of these models may expand their benefits. In addition, combining process-based and statistical or data-based models may lead to improved simulation of FIB in coastal waters (Hannides *et al.* 2021).

Decision support tools, such as EPA's Virtual Beach, are used to predict FIB concentrations at beaches and to inform beach closures and swimming advisories but can also be useful for researchers and engineers who are interested in relating FIB to environmental factors (Cyterski *et al.* 2013). This predictive model is used to estimate FIB based on independent variables such as water temperature, turbidity, and specific conductance. The use of nowcast systems has also been a highly useful but site-specific management tool, which provides water quality predictions on a daily basis, and are very useful for increasing public access to water quality data and beach conditions (Searcy *et al.* 2018; Boehm *et al.* 2007).

Public education & access to data

The first step in getting more water quality information in the hands of the public is to raise awareness about water quality issues. Increased awareness of water quality hazards as well as increased monitoring at recreational sites not qualifying for federal BEACH Act funding are two public health strategies recommended by the CDC (Graciaa *et al.* 2018). Increased public awareness may also help with illness reporting, as people do not necessarily associate illness with water recreation because symptoms can develop days after exposure (Craun *et al.* 2005; Esschert *et al.* 2020). Additionally, not all members of the public are equal stakeholders in recreational water quality. Access to recreational space is not equitably distributed, and there are disparities between communities with respect to water safety (Rigolon 2016; Gilchrist and Parker 2014).

Education is important for growing public support for water quality improvement projects. Keeping fecal pollution out of waterways requires a watershed-wide approach where a wide variety of infrastructure projects are implemented.

Nutrients and harmful algal blooms

Lead co-author: Martha Sutula

Harmful algal blooms (HABs) are a global environmental threat (Brooks *et al.* 2017; Anderson *et al.* 2021) accelerating with global change (IPCC 2019). HABs are defined as blooms of cyanobacteria, macroalgae and/or eukaryotic algae having a negative consequence to society or ecosystems. When conditions are favorable for certain species, they rapidly reproduce and accumulate biomass (Paerl *et al.* 2016). The accumulation of high biomass causes eutrophication (Nixon 1995), resulting in a cascade of problems. They can produce toxins causing illness and death in humans, domestic animals, and wildlife. HABs reduce aesthetics and cause taste and odor problems. HABs can cause low dissolved oxygen, acidification, reduced water clarity, poor quality benthic habitat, and acute and chronic impacts of toxins, all of which reduce the biodiversity and productivity of our coastal ecosystems. These conditions impact multiple human uses including drinking water, recreation, navigation, commercial and recreational fishing, and tribal and cultural uses (Griffith and Gobler 2020). Human activities are altering the environment in ways which promote HABs. Nutrient pollution is one major cause, but other factors such as hydromodification, physical habitat alteration, and organic matter dumping or sewage spills also contribute (Paerl *et al.* 2016).

The fundamental challenge with HABs is the pace and severity of outbreaks is not matched by science, monitoring, and support for communities to identify and implement solutions. The impact to quality of life, coastal economies, cultures, and coastal ecosystems is linked, but has not been properly quantified despite its severity. Substantial challenges exist from aging infrastructure, inconsistent or nonexistent monitoring and well-targeted mitigation, and lack of buy-in from communities and governments to fund monitoring, infrastructure upgrades, and direct and indirect mitigation resulting in meaningful solutions to HABs. Here, we briefly outline the science and coastal processes research contributing to communities' ability to understand, forecast, and mitigate HAB events, and the societal actions and funding needed to improve management.

The connection between HABs, their impacts on ecosystem services, and their environmental drivers are critical lines of investigation, but these linkages have not been systematically evaluated in most coastal regions. To meet this challenge, research is needed in five main areas: 1) monitoring technologies, 2) impacts on human health and aquatic life, 3) socio-economic effects, 4) environmental drivers, and 5) HAB mitigation approaches. To address the threat of HABs and its main root cause — nutrient pollution — societal needs have been classified with three primary categories: 1) outreach and education, 2) early warning and event response, and 3) possible next steps.

SOCIETAL NEEDS AND TOOLS/ TECHNOLOGIES NEEDED TO ADDRESS THEM

Monitoring technologies

Monitoring is logistically challenging and expensive because HABs are ephemeral, occurring far from their drivers with a manifestation of symptoms highly dependent on the waterbody's intrinsic factors. High frequency monitoring can help protect public health and investigate drivers. Participation of volunteers to collect, and in some cases, process field samples can greatly reduce costs and many states are now harnessing the leveraging power of citizen scientists. Numerous citizen science initiatives are conducted in collaboration with government agencies, such as CyanoScope, a bloom-monitoring program in Lake Superior in collaboration with Lakehead University in Thunder Bay, Ontario, and the EPA's Great Lakes Toxicology and Ecology Division, and the HABscope, used for *Karenia brevis* monitoring (Hardison *et al.* 2019).

Cost-effective, precise, and accurate monitoring technologies employable by trained citizen science groups would expand the amount and quality of HAB monitoring (Smith *et al.* 2021). Molecular methods are now mainstream (Bush *et al.* 2019) and can be incorporated into citizen monitoring, providing rapid, affordable, and high-quality data on algal community structure, HAB species and toxins (Medlin 2013). Expansion of the DNA reference library, bioinformatic pipelines, and metagenomic methods are needed to quantify HAB cell abundance, toxin concentrations, and environmental triggers (Medlin and Orozco 2017; Zhang and Zhang 2015). The "lab in the suitcase" approach is desirable (Acharya *et al.* 2020), where rapid field analyses can produce immediate action to protect public health.

Remote sensing methods have the potential to provide high frequency information on HABs (Stumpf and Tomlinson 2005; Shen *et al.* 2012; Hill *et al.* 2020; Cao and Han 2021); however, limitations on interpretation of remote sensing data in coastal environments remain (Nezlin *et al.* 2007). High resolution, hyper-spectral imagery are needed, as well as the routine processing that provides low-cost imagery to a wide range of end users (Wolny *et al.* 2020).

Finally, *in situ* sensor technologies are rapidly evolving to provide high frequency acquisition of HAB, nutrient, dissolved oxygen, and pH data. However, the sensor technology is typically high cost and beyond the reach of many citizen-science groups. Decreases in costs and operational complexities will yield more consistent, high-quality, and informative monitoring data.

Risk to human and ecosystem health

Multiple pathways exist for HABs to impact humans and the environment. While a basic understanding of the acute effects of toxins on human and animal health exists, multiple exposure pathways, threats of bioaccumulation, interactive/additive effects of multiple toxins, and effects of chronic exposures remain understudied areas. Guidelines for consumption and exposure are usually established for the typical use case, so certain populations (subsistence fishing, tribal and cultural uses) can be at greater risk due to high and/or chronic concentrations associated with more intense and complex exposure pathways (Smith *et al.* 2021). These uses can proceed even if a HAB is present and, therefore, research is needed to quantify guidelines for setting limits for chronic and complex exposure to cyanotoxins.

The data gaps are even more significant for pathways of exposure and risk to aquatic life, including marine mammals, migratory and resident birds, and other protected species. Though marine HABs dominate in coastal areas, multiple studies have documented inland cyanobacterial blooms can impact aquatic life in downstream coastal systems (e.g. death of endangered sea otters; Miller *et al.* 2010; Kudela 2011). More research is needed to understand how complex mixtures of multiple toxins and chronic exposure can bioaccumulate and adversely affect aquatic life, including physiological, behavioral, and even transgenerational effects. Effort is also needed to understand how other contaminants interact with toxins and other eutrophication stressors (DO, pH) to adversely impact organismal fitness.

Societal impacts and costs

The socio-economic and cultural impacts of HABs are severe, including impacts to public health (Backer and Moore 2010), commercial fisheries and aquaculture, recreation and tourism, home values and commercial real estate (D'Anglada *et al.* 2018), as well as disruption to social and cultural practices (Willis *et al.* 2018). A single major HAB event can cost local coastal economies tens of millions of dollars.

A 2018 study conducted by economists for Environment and Climate Change Canada (Smith *et al.* 2019) estimates algal blooms will cost the Lake Erie basin $272 million annually over a 30-year period if they continue at their current rate. Most of those costs are attributed to loss of tourism ($110 million annually), and the impact on residents' loss of recreational activities and lifestyle.

HAB impacts compound other issues in economically disadvantaged communities, such as limited access to recreational opportunities, clean water, health care and safe housing. Regionally specific studies are needed to better understand the magnitude of socio-economic impacts and provide timely information to decision-makers in order to motivate action.

Environmental drivers of HABs and eutrophication

HABs and other eutrophication symptoms occur as a consequence of environmental factors (aka drivers), including ample supply of nutrients, calm and stratified water, irradiance, and warm temperatures (Paerl *et al.* 2016). The processes driving variations in many of these characteristics are not understood well. Research is needed to understand better transport processes in river plumes and tidal inlet flows, overland flows during heavy rain or coastal storm inundation, wave driven currents, and coastal and beach groundwater effects on nutrient inputs and stratification. Mixing, owing to wind and waves, also can affect stratification, water temperatures, and nutrient transport, but is not understood well in shallow coastal and estuarine waters.

Coastal HABs are linked with excessive nitrogen (N) loading (Ahn *et al.* 2011), while in the Great Lakes phosphorus (P) is the focus. Consensus exists that controlling of both N and P is needed (Paerl and Otten 2013). Micronutrients and trace metals influence phytoplankton community structure and are important to consider. Hydromodification, shoreline hardening, floods and fires, and removal of riparian habitat are some factors contributing to HABs. Climate change is exacerbating HABs (IPCC 2019) because it enhances the specific environmental drivers promoting their growth (i.e. increased temperature, atmospheric pCO_2, irradiance, hydromodification; Burford *et al.* 2020). While these general drivers of HABs are well described, drivers influencing the specific HAB species blooming in a given waterbody, the exact timing, duration and location of a bloom, and the factors eliciting toxin production are still not well understood for most waterbodies. Thus, given the current state of research, predicting blooms requires a site-specific toolkit of observations, models and supporting research to be able to disentangle drivers, identify system interconnectivity, and specify the nutrient loading and flow requirements of coastal habitats, which will vary along the coast. Better understanding of the processes and feedbacks and interactions between drivers may result in an improved system understanding and applicability of larger scale models and observations.

Predictive models are a fundamental component of this toolkit (Burford *et al.* 2020). Models can be statistical or numerical but must be mechanistic to identify causal linkages to drivers. Model validation is essential for management confidence to apply them for decision support. Observations or watershed loading models are used to predict flows and nutrient loading from local contributing basins to receiving waters. Coastal hydrodynamic and (biogeochemical) water quality models incorporate forcing from the atmosphere, watershed, and the ocean to predict spatially explicit mass balances of oxygen, nutrients, and carbon, including contributions from various primary producer functional groups. Numerical models are advantageous because scenario analyses can be used to attribute sources and test management options to better understand the tradeoffs between flow, water quality, and HABS, and how management actions (source reduction, treatment versus ecosystem restoration) can buy increased ecosystem resilience. Open source and community supported models are especially beneficial as they allow refinement by academic research (Sutula *et al.* 2021).

While coastal hydrodynamic and water quality models predicting algal biomass, DO and pH are in routine use, predicting toxic HABs events from waterbody hydrodynamics and water quality is still an emerging and rapidly evolving area of science (Burford *et al.* 2020). Insufficient observations and experimental data have hampered the development of robust, mechanistic models representing first principles of planktonic life cycles and physiology. Hindcasts and seasonal forecasting based on proxies of HAB biomass is most advanced for well-studied systems (e.g. Lake Erie; Obenour *et al.* 2014), but the science of predicting toxic events at a whole-waterbody scale is in its infancy (Burford *et al.* 2020). Incremental steps are useful, e.g. empirical models can be used to refine regional or waterbody-specific risk relationships of probability of increasing toxic bloom events with temperature, nutrients or support short-term forecasts of HAB blooms (Wynne *et al.* 2018). The multi-agency multi-academic NOAA Coastal Coupling Community of Practice is an ongoing step toward linking hydrology inputs to the ocean, see also the Fate and Transport and Improved Modeling subsections of the Fecal Pollution section of this report.

Building a predictive HAB modeling toolkit requires a sustained long-term monitoring and research program, implemented in both the coastal system and its contributing watersheds. These observations need to link environmental drivers to eutrophication and HAB responses. Most states' existing HAB monitoring programs are focused on recreational health, so there is a need to expand monitoring to address drivers. If effectively synthesized, this evolving baseline of observations combined with existing water quality monitoring and modeling can fuel hypothesis testing, additional experiments and ultimately build an emerging scientific consensus on the major drivers for blooms and triggers for toxic events.

Mitigation of HABs and eutrophication

Coastal managers often have limited knowledge or resources to characterize and manage HABs. Most coastal habitats are either unstudied or have inadequate monitoring programs. Poorly chosen mitigation strategies yield a low chance of success and therefore result in lost revenue, an enduring problem, and long-lasting ecological damage. Short-term mitigation measures typically consist of treatments to reduce toxins or bloom biomass from the water column in order to quickly resume using the waterbody. Many examples exist, but some have environmentally damaging side effects (copper sulfate and other chemical algaecides, etc.). Research is needed on more benign approaches, examples of which include algaecides that self-degrade (e.g. peroxides), mechanical surface skimming and longer acting, non-toxic chemical treatments (e.g. alum, clay). More importantly, selection of the optimal method must be guided by understanding what is controlling HABs in a specific waterbody. Long-term HAB prevention requires a more thorough understanding of watershed forcing and *in situ* drivers contributing to blooms, including effects of top-down grazing. This better understanding would enable design of a cost-effective long-term mitigation strategy of watershed actions and on-site mitigation (e.g. dredging, sediment caps) to reduce the risk of HABs.

Decision support

Decision support tools include data management and visualization scripts/interfaces. Two categories are needed: 1) public health protection; and 2) HAB water quality management, which generally encompasses decisions on prioritizing ambient monitoring, regulatory actions, causal assessment, and mitigation and actions to conserve or prevent degradation of habitat. Stakeholders have explicitly called out the difficulty in using current data management systems to inform actions, be it avoiding a recreational beach, or listing a waterbody for impaired uses (Smith *et al.* 2021). A decision support tool kit is needed to guide managers towards cost-effective and environmentally acceptable mitigation approaches. Investments in systems supporting open data, improved data management, analyses and visualization are needed to protect public health and better respond to the threat of HABs. Central to this is making data and post-processing scripts freely accessible, usable, and shareable. Effective data management considers the entire data life cycle and requires developing systems to address each data life cycle stage — collecting, processing, storing, analyzing, interpreting, and accessibility.

EDUCATION AND POLICY

Public outreach and education

Protecting public health and addressing the causes and impacts of HABs cannot happen without public education and awareness of the threat as well as support and participation in identifying and implementing solutions. The public needs to have timely, easy to understand information to protect themselves, mobilize participation to address data gaps, and identify and implement solutions. Citizen monitoring is an essential component because the frequency of monitoring required to protect public health would make costs of such monitoring otherwise intractable.

Early warning and event response

Early warning and event response is needed to address the human health risks associated with exposure to HABs and their toxins in drinking water, at recreational use sites, and for seafood and shellfish consumption. Event response should also include marine mammal and wildlife centers, who respond and treat stranded animals exposed to HABs. Response actions are typically triggered by monitoring of a large bloom, observations of a suspicious scum, reports of illness, or mortality events. When available, early warning is provided by monitoring or predictive models that can give advance notice on the scales of days to weeks prior to a HAB event.

The fundamental challenge not currently being met is the public and agency need for timely and easily understood information on HAB events. To some extent, every U.S. state and territory monitors HABs and shares information with the public (EPA n.d. a); however, limitations exist on the extent and frequency of monitoring and the speed and accessibility of data sharing. Moreover, HAB notifications are often siloed, rather than consolidated with other recreational health or shellfish advisories. New Zealand's Land Air Water Aotearoa (LAWA n.d.) is an example of a centralized environmental data management system consolidating all available and relevant water quality information related to human health risks at a given site.

Following HAB events, investigation of drivers as well as implementation of remediation strategies is important. Typically, action lags because the event response data may not fulfill requirements to act (i.e. sufficient to place on a 303(d) list for impaired waterbodies).

Possible next steps

Addressing the problem of HABs and eutrophication requires: 1) reducing the threat to human and wildlife health, so impacted uses can be resumed and 2) taking steps to reduce the risk of HABs by understanding and addressing their root causes. Collectively, this is referred to as "mitigation." The most successful mitigation approaches are community-based, to address the local conditions and human activities contributing to HABs. Short-term mitigation measures include methods of reducing blooms and water column toxins in order to make the water safe to swim in, drink, and safely consume the fish and shellfish.

Addressing the root causes of HABs requires understanding the drivers, which are often site-specific. However, a major driver of HABs is anthropogenic nutrient loading, so nutrient management is a major thrust of HAB mitigation. Major sources include both point sources (e.g. municipal and industrial wastewater) and nonpoint sources (e.g. runoff from agriculture, confined animal feedlots, and urban infrastructure). Upgrades are needed to aging sanitary (wastewater treatment plants and septic systems) and stormwater infrastructure (separation from sanitary sewer infrastructure), install best management practices (EPA n.d. c) and implement low-impact development measures.

The restoration of coastal habitat will help reduce, for example, the risk of HABs. This includes establishing freshwater flow, removal of dikes and infrastructure impeding circulation, wetland restoration and installation of living shorelines, and removal of invasive species.

Microplastics

Lead co-author: Susanne Brander

Plastic pollution has emerged alongside climate change as a pressing global environmental problem. By 2040, it is expected plastic waste inputs will double relative to current levels based on recent trends, (Borrelle *et al.* 2020; Stubbins *et al.* 2021), and anthropogenic mass now exceeds living biomass globally (Elhacham *et al.* 2020). Plastics rapidly deteriorate physically into smaller particles but can take tens to thousands of years to chemically degrade, depending on their composition (Biber *et al.* 2019; Chamas *et al.* 2020). This may also be true of some types of bioplastics, which are marketed as a greener alternative, particularly in aquatic or marine ecosystems where they take longer to degrade (Carteny and Blust 2021). Due to the increasing production of plastics and mismanagement of plastic waste over the past seven decades, plastic particles are detected ubiquitously across aquatic and terrestrial ecosystems and ingested by biota (Brahney *et al.* 2020; Geyer *et al.* 2017; Rillig 2012). It is estimated that greater than 80% of marine plastic pollution comes from land-based sources, with microplastics contributing nearly one million tons (Jambeck *et al.* 2015). The majority of microplastics eventually sink to the seafloor (94%), with the remaining 6% being deposited on beaches (5%) or remaining at or near the sea surface (1%) (Carney Almroth & Egger, 2019; Pabortsava & Lampitt 2020).

Microplastics (MPs) present risks to coastal ecosystems due to their ability to be directly ingested or trophically transferred, physically damaging tissues after ingestion or aspiration, inhibiting assimilation of nutrients in the gut, translocating within organisms, leaching chemical additives, harboring infectious pathogens, and altering bacterial communities (Bakir *et al.* 2014; Chen *et al.* 2019; Cole *et al.* 2015; Wu *et al.* 2019; Athey *et al.* 2020; Seeley *et al.* 2020). MPs have been observed in biota at all levels of the food chain, from primary producers to top predators, including sources of seafood, and humans (Baechler *et al.* 2019; Zarus *et al.* 2021). Rather than being one uniformly identifiable pollutant, MPs are a suite of contaminants with a diverse range of shapes (e.g. fibers, fragments), sizes, polymer types, and associated additives or sorbed pollutants

(Rochman *et al.* 2019). Microplastics' complexities challenge assessments of risk (Koelmans *et al.* 2017), however, recent analyses estimate ecological risk thresholds have already been exceeded for ~1.5% of global surface waters (Koelmans *et al.* 2020), with predicted exponential and irreversible increases barring significant global intervention strategies (Everaert *et al.* 2020). Coastal ecosystems exist at the interface between marine, terrestrial, and riverine ecosystems, and thus plastic litter generated on land is transported by wind and rivers into nearshore environments (Lloret *et al.* 2021). Additionally, coastlines and estuaries tend to have high levels of urbanization, making them particularly susceptible to MP pollution (Gray *et al.* 2018; Weinstein *et al.* 2016).

The number of publications related to MP pollution has increased exponentially over the past decade (Granek *et al.* 2020). More research is needed to understand MP implications for coastal ecosystems, for example, significant data gaps remain regarding fate and transport processes, source identification, and health effects in wildlife and humans. A challenge throughout all these areas is the need for standardization in the way plastics are measured across environmental matrices (water, sediment, tissues) and better alignment in how toxicity experiments are designed and reported (Brander *et al.* 2020; Cowger *et al.* 2020). Here, we review the state of the science and provide recommendations for research priorities.

SOCIETAL NEEDS & TOOLS/ TECHNOLOGIES NEEDED TO ADDRESS THEM

Baseline occurrence and source identification

MPs are found in water, sediment, air, and biota in freshwater, estuarine, marine, terrestrial, and atmospheric ecosystems globally (Cole *et al.* 2011; Lusher *et al.* 2020), and the ocean contains upwards of 5.25 trillion plastic particles weighing at least 268,940 tons (Jambeck *et al.* 2015). Primary MPs originate from personal care products and some paints, industrial disrupting agents such as sand-blasting media, and pre-production pellets (nurdles) (Andrady 2017). Secondary MPs are more commonly found in the marine environment (Brander *et al.* 2020), and are produced via the breakdown and weathering of larger plastic items or are generated from synthetic clothing and fishing gear (Gago *et al.* 2018; Lusher *et al.* 2018). Weathering of plastic litter on beaches may be the largest source of secondary MPs to coastal areas (Hidalgo-Ruz *et al.* 2012), although recent research points to roadways as being a considerable and potentially underestimated source to waterways globally, representing over 80% of airborne MPs (Brahney *et al.* 2021). Developing a global baseline for MP concentration in coastal environments is challenging due to inconsistent methodologies for sampling, processing, analytical characterization, and reporting (Bergmann *et al.* 2015), although efforts to move towards standardization are under way (Cowger *et al.* 2020).

Coastal areas with high urban land use downstream of major waterways are expected to have high levels of MP contamination (Su *et al.* 2020), with evidence along beaches and in other coastal ecosystem types such as mangroves (Barasarathi *et al.* 2014; Horn *et al.* 2019; Nor and Obbard 2014). While the highest concentration of MPs in coastal ecosystems can be found in sediments, a significant proportion of MPs remain buoyant for extended periods and thus remain suspended in the water column where they are more likely to interact with pelagic life (Song *et al.* 2018). MPs have been observed in the tissues of many coastal species including those frequently consumed by humans (Baechler *et al.* 2020; Rochman *et al.* 2015). Airborne MPs have also been detected in coastal areas, with concentrations decreasing with increasing distances from the coastline (Brahney *et al.* 2021; Liu *et al.* 2019). Although urban areas are a clear source of plastics, lighter MPs such as fibers are often detected in remote areas (Allen *et al.* 2019; Ross *et al.* 2021). Globally, the oceans have also become a significant transport mechanism for plastic pollution, as MPs are caught up in oceanic gyres and currents, as well as via delivery of MPs back to the atmosphere via sea spray and wave action (Allen *et al.* 2020; Brahney *et al.* 2021).

Synthetic fibers are one of the most abundant and widespread plastic shapes (Rochman *et al.* 2019) across environmental matrices, including global oceans and marine biota (Athey and Erdle 2021). Recent studies point to laundering practices (washing and drying of clothes) in developed countries as being responsible for an outsized portion of synthetic fibers, much of it polyester, in even remote locations such as the Arctic Circle (Ross *et al.* 2021). Another often overlooked source of MPs is solid waste, in the form of landfill refuse, biosolids from wastewater treatment plants applied to agricultural fields, and food waste (Golwala *et al.* 2021). Coastal habitats, such as estuaries, tend to accumulate plastic debris and may facilitate higher rates of degradation into MP-sized fragments or particles (Weinstein *et al.* 2016), but less is known about their fate and transport in these areas in comparison to the open ocean.

Fate & transport

The fate and transport of MPs are governed by the geophysical flows in the ocean and the atmosphere, as well as the physical, chemical, and biological properties of the MP particles. Studies on the transport of MPs in the ocean have primarily focused on open ocean transport of floating debris (van Sebille *et al.* 2015; van Sebille *et al.* 2020) and transport dynamics between the open and coastal oceans are not well understood. While research in this area has benefitted from a fundamental understanding of hydrodynamics and sediment transport (Elfrink and Baldock, 2002; Lentz & Fewings 2012; Masselink and Puleo 2006), MPs have unique combinations of sizes, shapes, buoyancies, and input pathways different from other particles in the ocean such as sediment, larvae, or oil droplets, and for these reasons necessitate targeted research.

One approach to analyzing MP transport in the ocean is to use a Lagrangian perspective following the trajectories of individual particles. Simulations of MP transport within an ocean circulation model use particle-tracking methodologies to track MPs as particles with specific properties such as buoyancy (Delandmeter and van Sebille 2019). Such combined hydrodynamic-Lagrangian particle-tracking models have been recently used to hindcast potential sources of stranded plastic litter in the Indian Ocean (Bouwman *et al.* 2016; Duhec *et al.* 2015), Aegean Sea (Politikos *et al.* 2017), and Adriatic Sea (Carlson *et al.* 2017). An inverse modeling approach which used both observed MP concentrations and particle tracking in the Mediterranean predicted high rates of MP beaching and sinking (Kaandorp *et al.* 2021). In this way, Lagrangian particle tracking and MPs observations together can be used to estimate sources and sinks of plastic along the coasts. A unique challenge of modelling MPs transport is their propensity to change over time due to degradation, fragmentation, and biofouling — which can alter their buoyancy (Kooi *et al.* 2017) — thus changing their position in the water column.

In the nearshore environment, coastal processes such as waves and tides play an important role in the transport and fate of MPs (Abolfathi *et al.* 2020; Ballent *et al.* 2013; Critchell *et al.* 2015; Critchell and Lambrechts 2016; Isobe *et al.* 2014a; Liubartseva *et al.* 2016; Vermeiren *et al.* 2016; Yoon *et al.* 2010). MPs enter coastal and marine ecosystems through riverine systems, coastlines, vessels, platforms, or even the atmosphere, eventually reaching beaches, tidal wetlands, and marine sediments, and may reside in the coastal environment or be exported to the open ocean via currents (Zhang 2017). Because ocean currents are normally stronger at the surface than at depth, the relative depth of MPs will affect their transport. Surface-trapped or floating plastic will be affected by the surface currents and windage, whereas particles mixed lower in the water column are subjected to reduced currents (Cohen *et al.* 2019; Zhang *et al.* 2020). This suggests MPs may be sorted based on their relative buoyancy and water column location, as was recently demonstrated in a simulation of MP transport in the San Francisco Bay estuary (Sutton *et al.* 2019).

The transport of buoyant MPs near the surface will also be influenced by surface gravity waves. Studies have suggested positively buoyant plastics are likely to be transported on-shore by the Stokes drift transport induced by waves, which refers to the average velocity of specific fluid parcels that can transport particles (Forsberg *et al.* 2020; Isobe *et al.* 2014b; Kerpen *et al.* 2020). In addition, research has revealed the shape (DiBenedetto *et al.* 2018), size, and buoyancy (Alsina *et al.* 2020; Calvert *et al.* 2021; Forsberg *et al.* 2020) of MPs can alter their transport in waves. Once on-shore, the residence time of plastic on beaches is governed primarily by swash-zone (upper beach between back-beach and surf zone) processes (Hinata *et al.* 2017). Coastal vegetation also can act as a MP trap (de Smit *et al.* 2021; Sanchez-Vidal *et al.* 2021), increasing the residence time of MPs onshore. MPs can degrade in the coastal zone through mechanical degradation and weathering (Kalogerakis *et al.* 2017; Efimova *et al.* 2018) and may ultimately be removed by biodegradation or photodegradation (Ward *et al.* 2019). As MPs degrade, their buoyancy may change, which will further couple their fate and transport. However, the relative timescales of these processes are not precisely known and could take decades depending on polymer type.

Transport of MPs under extreme events such as floods, tsunamis, and storms has been investigated only in a few studies (van Sebille *et al.* 2020). For example, Osinski *et al.* (2020) characterized MPs transport under a storm surge event in the Baltic Sea in 2019 and found only higher-density MPs (300 µm diameter) were transported. MP concentrations in coastal waters have also been observed to increase after storm events (Moore *et al.* 2002; Wang *et al.* 2019). These increases may be due to stormwater runoff, or due to changes in marine transport, and more work is needed to fully understand these pathways (Willis *et al.* 2017), particularly since stormwater is known to be one of the largest sources of MPs to coastal waterways (Miller *et al.* 2021; Sutton *et al.* 2019).

Observations in biota

In studies on marine organisms from a wide swath of taxa, some common trends are emerging in terms of responses to MPs. At the cellular and molecular level, generation of reactive oxygen species and related expression of antioxidant enzymes is apparent across organism groups, from algae through fishes and mammals; however, it is sometimes difficult to ascertain whether the presence of these markers indicates a protective or adaptive response, or rather are an indication of stress and increased potential for impacts on growth, reproduction or other endpoints of concern (Jacob *et al.* 2020). Evidence supporting the latter can be found in terms of histological damage to the digestive tract and liver, indicating tissues are sometimes afflicted following MP exposure (Ahrendt *et al.* 2020; Espinosa *et al.* 2019). At higher levels of biological organization, growth inhibition is demonstrated in many taxonomic groups (Koelmans *et al.* 2020). In algae, biomass is reduced possibly due to decreased photosynthesis in response to some MP exposures (Ripken *et al.* 2020; Rocha *et al.* 2020; Su *et al.* 2020), and in other organisms stunted growth could be due to food dilution or decreased nutrient absorption (Koelmans *et al.* 2020). Changes in behavior, in organisms capable of movement, are also common (Costa and Malafaia 2020). Alterations such as changes in swimming speed or direction, as well as changes to more complex behaviors like shoaling, have been observed in several studies (Jacob *et al.* 2020; Yin *et al.* 2018; Yin *et al.* 2019). There are also indications fibers may be more toxic than spheres or fragments (Jacob *et al.* 2020; Stienbarger *et al.* 2021).

Some of the changes described above could have ecosystem level implications, particularly in terms of altered behavior in ecosystem engineers, such as marine worms and bivalves (Boots *et al.* 2019; Green *et al.* 2019). At the base of marine food webs changes in the sinking rates of algal species may alter both the amount of organic material and the rate at which it is transported to the deeper ocean (Ripken *et al.* 2020). As opposed to other pollutant, long-term impacts of MPs are less well-studied and research estimating the possibility of ecosystem or community level damage is lacking. However, changes in reproduction are observed in response to some MP exposures across all groups of marine organisms (Jacob *et al.* 2020; Sussarellu *et al.* 2016), which could cumulatively impact population size and biomass over time. Some studies even point to the potential for multigenerational or transgenerational effects, although potential for adaptation to MP exposure is also possible (Zhang *et al.* 2019). While it is unlikely MPs are impacting fisheries at current concentrations, sensitivity varies widely (Everaert *et al.* 2018), and modeling approaches are critical to predicting where and when impacts may occur over time.

Standardization

Baseline monitoring of environmental contaminants is necessary to assess relative contributions from multiple sources, assess risks, and develop strategies to reduce contamination (Wyer *et al.* 2020). Science and regulatory agencies in the U.S. require the existence of a reproducible method for sampling and analysis with well-documented uses and limitations (i.e. standardized methods) to mandate monitoring or develop regulations for contaminants. Comparability of MP monitoring data has hampered environmental management efforts (e.g. Great Lakes, United States) and drinking water (e.g. Denmark), highlighting the urgent need for standardized methods (Twiss *et al.* 2016; Løkkegaard *et al.* 2017).

While scientific organizations have long called for the harmonization of analytical methods for MPs (Twiss 2016; Hidalgo-Ruz *et al.* 2012), standardization remains elusive. Impediments for standardization have been due in part both to technical and logistical challenges. One of the most challenging technical issues with the analysis of MPs deals with characterizing polymer composition of small particles. Two of the most common spectroscopic techniques for identifying particle composition — Raman and Fourier-Transform Infrared (FTIR) Spectroscopy -- can only be as accurate as their underlying spectral libraries allow (Primpke *et al.* 2020). The recent development of free, crowd-sourced spectral libraries focusing on environmentally relevant MPs has significantly improved the efficacy of spectroscopic identification techniques (Cowger *et al.* 2021). Furthermore, MPs occur in greatest abundance in the environment at sizes smaller than 20 micrometers (Kooi and Koelmans 2019), which requires the use of microscopy coupled to spectroscopic tools to characterize their presence. Due to MPs being a unique environmental contaminant (insoluble particles), there is a relatively low demand for such instrumentation in commercial analytical laboratories, which provide the majority of regulatory required monitoring. Furthermore, such instrumentation can be relatively expensive ($100,000-$400,000), require highly trained personnel, and require extensive time for sample processing if not automated (Primpke *et al.* 2020).

Standardized methods or not, sampling and analysis plans should be devised based on study objectives. For example, manta trawl sampling (~100-500 μm lower size limit) and analysis with FTIR-ATR or microscopy with Nile red is a relatively inexpensive, simple method for characterizing occurrence in marine surface waters over relatively large areas; however, this method would not elucidate sources, compositions, or the abundance of small particles capable of tissue translocation (~<75 μm) (Labbe *et al.* 2020; Brander *et al.* 2020; Jovanović *et al.* 2018). Extrapolation to smaller-sized MPs is possible using compartment-specific probability density functions, with a higher degree of accuracy obtained with site-specific measurements based on sampling with in-line filters or whole water grab samples, and analyzed using appropriate techniques (e.g. micro-FTIR, micro-Raman, scanning electron microscopy, etc.) (Kooi and Koelmans 2019; Koelmans *et al.* 2020).

To date, California is the first U.S. State to legally require the development of standardized methods for sampling and characterizing MPs to fulfill several legislative mandates (California Code of Regulations 2018a; California Code of Regulations 2018b), as well as being the first state to adopt a definition for "microplastics" as a contaminant suite. To address these legislative mandates, the State Water Resources Control Board, in collaboration with the Southern California Coastal Water Research Project, initiated an inter-laboratory method validation study with 40 participating laboratories to develop a standardized analytical method for MPs in drinking water, turbid water, sediment, and fish tissue with an anticipated completion date of December 2021 (Martindale *et al.* 2020). Additional entities have initiated or completed method harmonization efforts for MPs collection and analysis in drinking water and other aquatic matrices, including ASTM International (ASTM WK67565; ASTM WK67788; ASTM D8332-20), the Joint Research Centre (European Commission 2018), and Japan (Michida *et al.* 2019). As analytical methods become standardized, selection of fit-for-purpose methods will be critical to ensuring data obtained are useful for the study objective (Coffin *et al.* 2021).

Policy and infrastructure

Plastic pollution is a challenge requiring systemic change and creative sustainable solutions (Lau *et al.* 2020; Stanton *et al.* 2020; Granek *et al.* 2020). Removing larger plastic items from already-polluted ecosystems is a common, albeit inefficient, approach to mitigating contamination (e.g. beach cleanups, mechanical trash removers). Once plastic has degraded into small particles, retrieval from the environment without causing harm to marine organisms and associated habitat is implausible (Hohn *et al.* 2020). The approaches needed to reduce plastic production are numerous and complex in nature. Ultimately, incentives enticing industry to transition towards reusable or biodegradable materials, with regulations to mandate the amount of plastics produced, and thus MP loading into marine and aquatic ecosystems, may be the most viable approach (Wyer *et al.* 2020; Brander *et al.* 2021). Circular economy approaches requiring companies to be responsible for their own waste treatment or recycling are popular. In 2021, the State of Oregon enacted a law holding producers of plastic, paper, and other materials responsible for funding educational programs and upgrades to recycling facilities, or other activities aimed at recovering these materials, starting in 2025 (Oregon SB582), but similar initiatives have not yet gained traction in the rest of the U.S. (Syberg *et al.* 2021). Plans for source reduction in the form of requirements to capture microfibers from washers and dryers, better mitigation of stormwater (a large source of MPs), and regulations prohibiting the sale of unnecessary single-use items (e.g. straws, shopping bags, etc.) are increasing in popularity but require enforcement to facilitate change. States such as California are in the process of working towards mandated monitoring of plastics in drinking and ambient waters (Coffin *et al.* 2021), and at the federal level bills such as the "Break Free from Plastic Pollution" Act have been proposed but not yet put into action (S.984 2021). Momentum is building to support actions oriented towards reduced reliance on plastics, but economic incentives and regulations are necessary to make this fully possible.

NEEDS AND RECOMMENDATIONS FOR COASTAL POLLUTANTS

Recommendations for scientific research, infrastructure needs, community engagement, and policies aim to help prioritize future research and investments. The USCRP is specifically interested in how a better understanding of coastal physical processes interactions with coastal pollutants (e.g. fate and transport, circulation, depositional environment) may benefit the following.

Observations and research

• Quantify potential threats to human and ecosystem health through accurate risk assessments.

• Determine where point and non-point sources of pollution are entering the environment through source identification studies.

• Develop lower cost and rapid response tools to help coastal managers better respond to pollutant threats.

• Conduct mitigation technique analysis and long-term studies to determine success over time, including increased sampling and statistical analysis of routine data.

• Improve pollutant impact forecasting by integrating frequent and new data points into existing and novel models.

• Account for the total cost of pollutants to society at-large, in terms of both current and future impacts on the economy.

Infrastructure, policies, and outreach

• Establish early warning and event response thresholds based on sound science to protect wildlife and human health.

• Empower users through effective public awareness campaigns to improve public access to data, with the publication of open data by researchers encouraged where possible.

• Develop additional decision support tools and training.

• Educate that restoration of physical habitat and hydrologic conditions can reduce pollutant loading (e.g. removal of dikes).

• Encourage water treatment facilities improvements, recycling, reuse, source reduction, low impact development, and agricultural load reduction

RESILIENT COASTAL ECOSYSTEMS

Coastal fisheries

Lead co-author: Jacob P. Kritzer

Coastal waters are among the most ecologically rich ecosystems in the world due to the enhancement of oceanic productivity by nutrients and sediment from freshwater and terrestrial sources. Furthermore, shallow depth enables more light penetration to the sea floor, promoting photosynthetic activity and greater variety, density, and complexity of vegetated habitats than is possible farther offshore. These conditions create productive and valuable fisheries in the coastal zone. At the same time, ease of access to these resources creates greater risk of overfishing. Overfishing is exacerbated by proximity to other anthropogenic impacts. Management of coastal fisheries increasingly calls for science and policies to account for the broader environmental context in which they take place.

Coastal fisheries typically harvest portfolios of species composed of both finfish and invertebrates. Although many environmental factors are important to both taxonomic groups, broad life history differences have important implications for the priority processes affecting each. Finfish are more motile and utilize coastal habitats in different ways at particular times, whereas invertebrates are generally more sessile and resident in these habitats year-round. Also, many invertebrates are shell-forming and some are filter-feeders, which are important determinants of their interactions with the environment. These are, of course, generalizations not applicable to all species in either group. Nevertheless, these attributes are sufficiently widely shared as to be informative for prioritization and planning.

This section focuses on finfish fisheries in the coastal zone, considering three broad life history types. First, some species use shallow, structured coastal habitats as nursery habitats before moving offshore as they grow. Spawning might also take place in coastal areas but can also take place offshore with young moving inshore to settle along the coast. Atlantic cod, for example, use deeper gravel and cobble beds as nursery habitat, but also shallow eelgrass meadows. Second, diadromous species by necessity pass through coastal waters in the process of spawning migrations between freshwater systems and the sea. Diadromous species like salmon and river herring are anadromous, meaning they live at sea but spawn in freshwater, whereas catadromous eels and other species live in freshwater but migrate to sea for spawning. Third, coastal migrants exhibit longshore movements with forays into different estuaries and embayments, often for feeding. Striped bass exhibit such migrations and are tracked by anglers to anticipate arrival of fish in different locations. Different coastal processes have greater importance for each of these broad categories of habitat use.

SOCIETAL NEEDS & TOOLS/ TECHNOLOGIES NEEDED TO ADDRESS THEM

Observations

Traditional fisheries science and management approaches call for data on the basic life history parameters for the exploited species (growth, natural mortality, maturity, fecundity), estimates of total harvest by all sources (i.e. commercial, recreational, and indigenous fleets; directed and non-target catch), and fishery-independent indices of abundance (Hilborn and Walters 1992). For coastal fisheries, inclusion of environmental data is also important given that the influence of non-fishing impacts on habitat and water quality are greatest in the coastal zone. For example, dynamics of Australia's Northern Prawn Fishery are strongly driven by rainfall, which has been incorporated into modeling frameworks (Dambacher *et al.* 2015). Trade-offs inherent in the costs of data collection and complexity of models can call for difficult decisions about sacrificing biological data collection for environmental data collection, or vice versa.

Certain environmental variables are important to all finfish species supporting coastal fisheries. Temperature and salinity determine whether coastal waters are within metabolic tolerances, and dissolved oxygen is critical for respiration, growth, and survival. Beyond these core variables, others will be more or less important to different species depending upon how each uses the coastal zone. Priority variables will vary from species to species, but broad differences among the three major coastal zone use types are as follows:

Coastal nurseries: Young-of-year and juvenile finfish typically rely on complex structured habitats as shelter from predators. In the coastal zone, these habitats are often biogenic, including submerged aquatic vegetation (SAV), macroalgae (i.e. seaweeds), and shellfish beds and reefs. As living organisms in their own right, the abundance and quality of these habitats will depend upon whether the prevailing environmental conditions fall within their range of tolerance. The most important factors can vary among habitat-forming species, with some having very different responses to the same conditions. For example, intertidal marshes rely on a steady supply of sediment for repair, expansion, and migration (Kennish 2001), whereas heavy sediment loads can be fatal to many bivalve shellfish (Wilber and Clarke 2001). Seagrasses have intermediate tolerance, requiring sufficient sediments for root systems to take hold but are vulnerable to burial when sediment loads are heavy (Cabaço *et al.* 2008).

Observatories supporting management of nursery-dwelling finfish in the coastal zone should monitor key habitats directly, but also the environmental drivers of their present and future status to predict changes and mitigate adverse impacts. The priority variables are likely to vary among locations as a function of fisheries species, their habitat needs, and the key environmental drivers. Therefore, observing systems should be tailored to each unique coastal fishery based on critical consideration of its priorities. Notably, these observing activities might benefit fisheries operating farther offshore depending upon the nature of ontogenetic habitat shifts.

Diadromous species: In the course of moving between freshwater and marine ecosystems, the biggest impact faced by many diadromous species is barriers to migration (Verhelst *et al.* 2021). Dams, impassable culverts, and other physical structural barriers have resulted in dramatic decreases in access to upstream spawning habitat for many anadromous species and upstream nursery habitat for many catadromous species (Hall *et al.* 2010, 2012). Installation of fish passage structures have mitigated the impacts of physical barriers to a degree, but many were engineered for flow conditions that are changing. Climate change increases variability in precipitation patterns (Balch *et al.* 2010), resulting in more flood and drought events. Diadromous species are confronting impassable high flow and low flow conditions as a result (Brown *et al.* 2010). Therefore, the most important observing priority for many diadromous species will be conditions for migration, particularly river flow as it interacts with physical barriers and fish passage structures. Physical or flow barriers in the coastal zone are especially important because they can restrict access to any freshwater habitat whatsoever, whereas spawning and nursery grounds might still be accessible below barriers fa rther upstream.

Notably, many anadromous species occupy coastal waters as juveniles after outmigration from freshwater habitats and before moving farther offshore as adults (Brown *et al.* 2000). During this stage, they can rely on structured habitats as nurseries and therefore are affected by changes in those habitats, as discussed above. Habitat conditions upstream of coastal areas in freshwater rivers, lakes, and ponds are also critical for all diadromous species, in addition to the effects of migratory barriers. Freshwater systems are among the most vulnerable by virtue of their restricted area and proximity to anthropogenic impacts. In fact, many marine fish species listed under the U.S. Endangered Species Act are diadromous given they are susceptible to cumulative impacts across freshwater, coastal, and offshore areas.

Coastal migrants: Some species of finfish range widely along coastlines, feeding at different locations on either predictable or opportunistic schedules. For example, striped bass, the most economically valuable sportfish on the U.S. Atlantic Coast, spawning in the Roanoke River can range over more than 1,000 km along the coast during non-spawning times (Callihan *et al.* 2015). Striped bass prey upon a variety of fishes and invertebrates utilizing coastal embayments and estuaries, either throughout their lives, for spawning, or in nurseries (Walters and Austen 2003). The abundance and residence time of migratory predators at any particular locale is likely to depend on the abundance of their prey, and therefore on the environmental conditions important to those prey species. Prey can include finfish using nursery habitats and diadromous species, so the variables important to those species as discussed above will also affect fisheries for coastal migrants.

Data

Environmental observing in the coastal zone entails unique advantages, but also important challenges. The proximity of human communities to the ecosystems makes it safer and more cost-effective to deploy observing systems, and the density of stakeholders presents more sources of financial, operational, and analytical resources to draw upon. At the same time, proximity and density also create a greater load of anthropogenic impacts complicating observing needs. Coastal areas are also typically much more complex in terms of geography, hydrology, and ecology than areas farther offshore, which limits the scales over which observing data can sufficiently characterize conditions and calls for higher resolution data collection.

These advantages and challenges apply to all environmental policy issues in the coastal zone but are exacerbated in the fisheries sector. Fishing fleets present valuable ships of opportunity for data collection (Gawarkiewicz and Mercer 2019), while the fishers themselves are unique reservoirs of local ecological knowledge (Silvano and Valbo-Jorgensen 2008). This means fisheries have greater potential for data generation than many other policy arenas. Data demands are also greater due to the number of species that are important to many fisheries, the complexity of their biology, and the interactions among them and with the rest of the ecosystem. It is, therefore, especially important for coastal fisheries to maximize the utility and efficiency of data systems. This means first conducting a thorough inventory of existing data streams, considering data priorities, before investing in new data collection to avoid redundancy. Making data publicly available wherever possible and promoting discoverability through centralized clearinghouses can expand the reach of existing data systems.

Modeling

Clearly identifying the models or analytical frameworks to be applied and how these feed into decision-making processes is essential to illuminating how science connects to policy and guiding data collection. The U.S. federal stock assessment enterprise provides an excellent model of these principles (Lynch *et al.* 2018), which could be replicated at the state and municipal levels who have jurisdiction over coastal fisheries. Key to this process is the clear regulatory response to stock assessment outcomes (science-based catch limits) that draws a clearer line from data collection through modeling to policy implementation (Miller *et al.* 2018). Although well-developed stock assessment systems like those in place for federally managed fisheries in the U.S. often do not consider habitat or environmental data (Caddy 2013; Tanaka 2019), these data can be readily incorporated into tools applicable in small-scale and data-limited contexts such as coastal fisheries (Honey *et al.* 2010). Environmental data can also be used in simulation-based framework such as Management Strategy Evaluation (MSE), which is increasingly used in complex systems characterized by high levels of uncertainty and interactions among fishing and non-fishing impacts (Harford *et al.* 2016). Environmental data can also be used in relatively simple semi-quantitative decision-making frameworks such as traffic light assessments to guide management of coastal fisheries (Caddy 2002; e.g. northern shrimp, Koeller *et al.* 2000).

Communication

Many fisheries around the world have improved communication among fishers, regulators, and other stakeholders through more participatory and cooperative scientific and management approaches. Capitalizing on fishing vessels as platforms for data collection not only expands the information base for management but can also build buy-in and improve relationships with the governance system (Conway and Pomeroy 2006). Some fisheries also go a step further in implementing co-management systems, sharing some or all of the decision-making responsibility with fishers. These approaches can empower fishers and further build buy-in to management (Jentoft 2005). In many coastal fisheries, co-management approaches are the only practical approach in light of the complex and disaggregated spatial structure of fishing fleets and fishery resources (Prince 2003). Although co-management systems are often viewed as a necessary tool in lower capacity contexts, in which resources are more limited for top-down governance, especially the developing tropics, there are notable examples of effective co-management systems in small-scale, data-limited fisheries in wealthier temperate settings. In the State of Maine, for example, co-management has helped to meet complex environmental challenges in fisheries for soft-shell clams (McClenachan *et al.* 2015a) and anadromous herring (McClenachan *et al.* 2015b).

Policy

Coastal fisheries face a wide range of challenges related to ecological complexity and uncertainty, scale, and interactions with non-fishing impacts. Meeting these challenges will be helped by an explicit accounting for uncertainty and change, aiming to build resilience and responsiveness into the governance system. Adaptive co-management approaches can tighten feedback loops between environmental change and management responses (Plummer 2009). Given the greater interactions among fishing and non-fishing impacts in the coastal zone, management frameworks could consider mechanisms for cross-sectoral decision-making. This can include formal requirements for consultation of other management authorities, such as the requirement for the Federal Energy Regulatory Commission to solicit input from Regional Fishery Management Councils in the U.S. However, fully accounting for impacts across ocean uses and developing solutions for optimizing outcomes across a portfolio of benefits can only be achieved through management systems which are truly integrated rather than siloed. The importance of non-fishing impacts in the coastal zone means that coastal fisheries are increasingly adopting approaches consistent with principles of Ecosystem-Based Fisheries Management (EBFM), but to account for and manage across a range of ocean uses, comprehensive Ecosystem-Based Management (EBM) is needed.

Shellfish

Lead co-author: Daphne Munroe

Shellfish are ubiquitous in estuaries and the coastal ocean. They support important sustainable human food production systems, aquaculture, and fisheries (Gephart *et al.* 2021), and play an important role in the health of coastal ecosystems (van der Schatte Olivier *et al.* 2018). Shellfish populations not only rely on healthy and clean waterways for habitat, they also contribute directly to cleaning and improving the productivity of those habitats. Through their filter feeding, shellfish help to clean water (Fulford *et al.* 2010; zu Ermgassen *et al.* 2013; Galimany *et al.* 2017), improving water clarity and supporting habitat for other ecologically important species such as seagrasses (Gagnon *et al.* 2020). By growing vertical structures such as oyster reefs, shellfish can also provide important habitats for other commercially and recreationally valuable species of finfish and invertebrates (Harding and Mann 2001; Lehnert and Allen 2002; Petersen *et al.* 2003; Luckenbach *et al.* 2016; Shinn 2021). The shellfish themselves, or the structures used to farm them, can further help to stabilize shorelines and protect them from erosion due to wave and sea level rise (Piazza *et al.* 2005; Scyphers *et al.* 2011; Pinsky *et al.* 2013). Collectively, the ecosystem and societal benefits furnished by shellfish have made their enhancement and restoration a priority (Grabowski *et al.* 2012).

As shellfish aquaculture continues to expand in the U.S., sustainable management of historically overfished shellfish fisheries is fostered, and efforts to restore and enhance shellfish populations amplifies, it is imperative the tools and technologies needed to ensure their success are developed and supported. A step towards developing and supporting the tools and technologies needed to support shellfish aquaculture, fisheries, and restoration is to identify and quantify what is known about threats to the health and resilience of shellfish populations. We will focus on four of the many identified as primary across shellfish aquaculture, fisheries, and restoration.

SOCIETAL NEEDS & TOOLS NEEDED TO ADDRESS THEM

Identifying and forecasting threats

Intense precipitation events in watersheds, also known as freshets, push low salinity, or in extreme cases freshwater, into coastal bays and estuaries where shellfish reside. These shellfish are benthic and unable to move to escape these weather events, often leading to large mortality events (Levinton *et al.* 2011; Pollack *et al.* 2011; Munroe *et al.* 2013). Climate models predict precipitation (Najjar *et al.* 2000; Hayhoe *et al.* 2008) as well as the frequency of extreme storm events (Voynova and Sharp 2012; Wetz and Yoskowitz 2013) will increase in the northeastern U.S. (Karl *et al.* 1995; Allan and Soden 2008). Indeed, these expected increases in the frequency and intensity of freshwater events (i.e. precipitation) will continue to exacerbate episodes of low salinity in estuaries (Sanderson *et al.* 2019), and shellfish mortality events due to freshets are already on the rise. Extreme flooding along the Mississippi River, for example, caused the opening of the Bonnet Carre Spillway twice in 2019, releasing freshwater over the wild oyster grounds, causing massive mortality in the wild stocks decimating the fishery (Gledhill *et al.* 2020). Hurricane Harvey in 2017 also created a severe and prolonged freshet that caused a mass oyster mortality event in Galveston Bay (Du *et al.* 2021). Although freshets can be devastating, they are one of multiple climate change impacts affecting shellfish.

Climate change has caused sea level to rise globally, on average, between 11-16 cm from 1900 to 2000 (Hay *et al.* 2015). Forecasts of future sea level rise in the current century vary widely from 16 to 254 cm (Garner *et al.* 2018). Increasing sea level has already begun to negatively affect coastal shellfish habitats through erosional loss, changes in sediment deposition, and salt intrusion (Wells 2021). Rising seas have also brought salt water farther into estuaries, increasing the salinity of important oyster beds in the Delaware Bay (Ross *et al.* 2015) and Chesapeake Bay (Hilton *et al.* 2008). Although bivalve shellfish are osmoconformers, meaning they maintain internal salinity levels in balance with ambient water, changes in estuarine salinity due to sea level rise will have important consequences for farmed, fished, and restored shellfish alike (Pourmozaffar *et al.* 2020). As sea level rises, optimal habitat for various intertidal and subtidal bivalve species may shift or be lost altogether. For example, if oyster reef accretion (i.e. vertical growth) cannot keep pace with sea level rise, those beds may be lost. However, in a recent study of restored intertidal oyster reefs in North Carolina, Rodriquez *et al.* (2014) demonstrated when located in high growth areas, the restored beds can in fact keep pace with the current rate of sea level rise. Conversely, ribbed mussels, a species playing an important role in stabilizing marsh edges along the East Coast of the U.S., may not be as resilient to sea level rise, with projections of loss of over half of the mussel population in the Chesapeake Bay over the next three decades due to marsh erosion (Isdell *et al.* 2020).

Erosion and siltation by terrestrial deposits are becoming increasing issues in coastal habitats and estuaries (Bilotta and Brazier 2008). Additionally, dredging of navigation channels results in the release of suspended sediments known to negatively impact oyster reef structures (McFarland and Peddicord 1980; Wilbur and Clark 2010), as well as causing issues for clam, mussel, and oyster fishery stocks ranging from developmental concerns for larvae to growth impairment of adults (Davis and Hidu 1969; Hopkins and McKinney 1976; Bricelj and Malouf 1984; Emerson 1990; Wilbur and Clark 2001). Much of the existing research has tested for impacts to shellfish from suspended sediments using controlled and relatively acute laboratory experiments. In a prolonged (seven-day) experiment, simulating delivery of suspended sediments from dredging activity showed no negative effects on oyster survival nor growth (Suedel *et al.* 2015); however, in a mesocosm study, siltation was shown to lead to lower body weight relative to size in oysters (Colden and Lipcius 2015). Increased exposure to suspended sediments has also been related to an increase in the incidence of parasite infection in farmed oysters suspended in the water column (Clements *et al.* 2017) and reduced settlement of larval oysters to aquaculture collectors (Poirier *et al.* 2021). As suspended sediments settle out they can smother and kill farmed oysters (Comeau 2014), and even a thin layer of sediments on benthic substrate can prevent oyster larvae from attaching and settling on those surfaces (McKinney *et al.* 1976; Thomsen and McGlathery 2006; Kuykendall *et al.* 2015).

Climate change will also cause shifts in average and extreme environmental conditions in coastal systems (Doney *et al.* 2012). As a result, shellfish in those systems may become more exposed to stressful, or even non-viable, environmental conditions due to concurrent changes in multiple stressors such as temperature, salinity, ocean acidification (OA), and dissolved oxygen (DO) (Byrne and Przeslawski 2013; Reid *et al.* 2019a). Climate change will increase both temperature and OA and the combined effects of the two have been commonly studied in shellfish (Miller and Waldbusser 2016; Lesser 2016; Speights *et al.* 2017), with many studies focused on the response of reproduction or larval stages because those early life stages have the potential to be more susceptible to OA due to the chemistry of the larval shell (Wittmann and Pörtner 2013; Hendricks *et al.* 2010; Waldbusser *et al.* 2015). For example, aragonite (a carbonate mineral necessary for shell formation) undersaturation, a consequence of ocean acidification, has been shown in laboratory experiments to reduce survival and growth of shellfish larvae (Gobler *et al.* 2014; Gazeau *et al.* 2013; Talmage and Gobler 2010; Waldbusser *et al.* 2014) and cause failures in shellfish hatcheries in the Pacific (Barton *et al.* 2012). In coastal and estuarine systems, anthropogenic inputs of nutrients into rivers and estuaries and resultant algal blooms generate concurrent stressful low DO and pH conditions (Baumann *et al.* 2015; Wallace *et al.*, 2014), making these a common multiple stressor for shellfish. Larval and juvenile scallops and clams, for example, are negatively impacted by either low DO or pH, but when experienced concurrently the consequences to survival and growth are enhanced (Gobler *et al.* 2014). Exposure to multiple stressors can also enhance disease susceptibility in wild (Lenihan *et al.* 1999; Volety 2008; MacKenzie *et al.* 2014) and farmed (Clements *et al.* 2017; Lardies *et al.* 2017) shellfish populations, with important implications for restoration planning (Keppel *et al.* 2015). The combined effects of multiple stressors on shellfish are complex, with some studies showing certain species remaining unaffected, whilst others can be either positively or negatively impacted (Lemasson *et al.* 2018). The stressors themselves can interact in unexpected ways (Reid *et al.* 2019b) making the combined response difficult to anticipate.

Data and monitoring

In the face of these numerous threats to shellfish in coastal ecosystems, data collection and monitoring remain important to the health and sustainability of coastal regions. A variety of data resources are available to managers, researchers, and resource users for tracking the status and resilience of shellfish populations. In the aquaculture sector, for example, state and federal agencies track farm production through industry census and reporting (USDA 2019). Shellfish farmers also have important reasons to track diseases impacting their livestock; therefore, data about disease occurrence and prevalence provide an important tool towards biosecurity for their crops (Rutgers University n.d.). For federal- and state-managed shellfish stocks, agencies regularly perform surveys to collect data on the status of the populations, to make management decisions. Shellfish fishery survey data are shared through stock assessment reports such as the New Jersey Oyster Stock Assessment Workshop Report (Haskin Shellfish Laboratory 2021), the Virginia Oyster Stock Assessment, and Replenishment Archive (Virginia Institute of Marine Science (VIMS) 2021), and federal assessments for Atlantic Sea scallops, Atlantic surfclams, and ocean quahogs (Northeast Fisheries Science Center n.d.), to name a few. Restoration projects often have associated monitoring and data collection efforts supplying useful information about the status and trends of shellfish in coastal habitats. Radabaugh *et al.* (2019) provided an example of a well-integrated resource, describing available data and programs for shellfish habitat and restoration monitoring in Florida. An important aspect of shellfish management for both farms and wild resources is water quality monitoring for harmful algal blooms and other human health associated water quality standards (NSSP 2019). Federal, state, tribal agencies, and other entities regularly collect water quality samples to define areas from which it is safe to harvest shellfish for human consumption. The USGS, EPA, and the National Water Quality Monitoring Council (NWQMC) jointly maintain a Water Quality Portal serving these data (EPA n.d. b).

Modeling

Water quality and other data regarding status and trends in shellfish populations can be used in a variety of ways. One way these data are used is to help parameterize models, which support policy and management decisions. Spatial models to aid in shellfish aquaculture planning and siting are increasing in use. Two examples are the shellfish aquaculture siting tool provided by the Alabama Marine Resource Division (n.d.), and an aquaculture mapping atlas jointly provided by the University of Connecticut, the Connecticut Sea Grant Program, and the Connecticut Department of Agriculture, Bureau of Aquaculture (2018). These, and other similar spatial tools, are intended to support data-informed decisions about where to site new farms in coastal areas. Models informed by shellfish data also include those used for basic research to understand the intersection of coastal processes and the health of shellfish populations that support fisheries. One such model used an oyster population dynamics model with a hydrodynamic model and an oyster mortality time series to better understand the impacts of extreme precipitation events on oyster mass mortality events (Munroe *et al.* 2013). Another study used a coupled hydrodynamic, water quality, and oyster population model to investigate the relative impacts of Mississippi River diversion and sea level rise on oyster growth and survival (Wang *et al.* 2017). Their study showed that large scale freshwater diversions had greater adverse impact on oysters than the impacts of sea level rise; however, small-scale diversions impacted oysters less negatively than sea level rise (Wang *et al.* 2017), highlighting the complex nature of the tradeoffs that face coastal resource managers. Models are also used to estimate water quality benefits of shellfish restoration efforts (Kellogg *et al.* 2018). For example, the Harris Creek shellfish restoration model includes an online simulation tool allowing users to explore the water quality, ecosystem, and economic benefits of various restored shellfish (VIMS 2018).

Natural and nature based features

Lead co-author: Curt Storlazzi

Coastal communities and ecosystems are threatened by a range of coastal hazards including flooding during storms, inundation due to sea level rise, shoreline change (erosion/accretion), decreased ice cover, and alteration of sediment dynamics due to changing climate or anthropogenic activities. Communities are increasingly looking for effective and suitable measures that can simultaneously mitigate coastal hazards, adapt to climate risks, reduce or reverse anthropogenic impacts to ecosystems, and contribute to coastal sustainability (Borsje *et al.* 2011; Temmerman *et al.* 2013). Natural coastal habitats such as reefs, beaches/sand bars, bluffs, dunes, mangroves, wetlands, and submerged aquatic vegetation can provide an effective first line of defense against these flooding and erosion hazards (Reguero *et al.* 2018, 2021; Sun and Carson 2020). However, natural protection services may be compromised as coastal habitats are degraded or lost due to climate change, sea level rise, water level fluctuations, and encroaching coastal development (Arkema *et al.* 2013; Gittman *et al.* 2015; Quataert *et al.* 2015; Crosby *et al.* 2016; Narayan *et al.* 2017).

The restoration, construction, or conservation of coastal natural and nature-based features (NNBF) provide a potential pathway to accomplish the multiple goals of flood risk reduction, climate adaptation, increasing public access, and promoting ecosystem functions and services. However, measures to quantify their effectiveness at mitigating coastal hazards and adapting to climate impacts are still lacking. Due to the site-specific nature of these approaches, contractors and decision-makers often lack information on assessing their suitability in different environments. With groins, seawalls, bulkheads, revetments, and breakwaters being the traditional default approach over the years, and research on NNBF suitability and monitoring only picking up over recent years, this growing field of information is yet to be institutionalized into policy, regulations, and decision-making. The development of risk-based valuations of the ecosystem services attributed to NNBF has been limited by the lack of high-resolution data on bathymetry, topography, ecosystems, and economic assets, and the difficulty in modeling complex hydrodynamic and ecological processes (Reguero *et al.* 2021). A better understanding of the hydrodynamics, morphodynamics, and ecology across a variety of spatial and temporal scales, as well as across a range of types of NNBF, is critical for clearly defining the benefits of NNBF and advancing NNBF-based solutions. In many cases, these benefits are not easily quantified. In such instances, traditional ecological knowledge collected and passed on via indigenous stewards of the land are important sources of information. Similarly, more recent experiences of coastal residents and landowners may provide useful data on NNBF outcomes and performance (Smith and Scyphers 2019; Smith *et al.* 2017).

SOCIETAL NEEDS AND MANAGEMENT ACTIONS TO ADDRESS THEM

Observations

Modeling and monitoring NNBF to characterize risk reduction performance across a large range of coastal environments, from the Arctic to the tropics, open-ocean coastlines to estuaries and the Great Lakes are a critical need. Observations of NNBF performance are generally quantified as the impact of the NNBF on (1) wave energy attenuation and the associated wave-driven water levels and coastal flooding, and (2) circulation and the resulting sediment, nutrient, contaminant, and larval dynamics. For example, how does wave energy decrease across the NNBF feature? Does the NNBF enhance sediment deposition along the shoreline and larval recruitment, or does it result in the concentration of nutrients and contaminants that negatively impact the associated ecosystems?

Monitoring of certain NNBF, such as marsh restoration projects, tends to be required under certain regulations, but is often limited to ecological functions (e.g. recruitment, growth rates, diversity) and doesn't cover a long timeframe adequate for assessing the performance of the project. Some states and NGOs are developing standardized monitoring metrics for NNBF, but funding to support monitoring is still an obstacle. More widespread, long-term, and consistent monitoring of the risk reduction performance of NNBF is needed to help inform management decisions and development of future design standards and inputs for hydrodynamic modeling.

Emerging observation technology may provide new data sources for NNBF monitoring. For example, unoccupied aerial vehicles (UAV) and satellites are useful for extensive and long-term mapping, whereas affordable, off-the-shelf sensors and loggers that can monitor ecological variables such as water temperature and dissolved oxygen are becoming increasingly available and can be deployed in networks to monitor ecosystems (Mao *et al.* 2019). Crowdsourcing and community science also provide innovative and scalable methods of NNBF monitoring (Harley *et al.* 2019). This includes methods such as photo-monitoring, beach profiling, and desktop-based characterization of imagery. With artificial intelligence (AI) methods becoming mature, data quality and quantity are expected to increase rapidly in the near future.

Modeling

Modeling of coastal NNBF needs to cover both the representation of the influence of the NNBF on coastal processes, such as wave and surge mitigation, and the sustainability of the ecological and morphological processes operating within the NNBF itself. With regard to influence on coastal processes, the key issue for representing most NNBF within wider two- and three-dimensional coastal hydrodynamical modeling is a better understanding of (a) the hydrodynamic roughness of the NNBF, and (b) the effects of NNBF on circulation and the resulting sediment transport. The effect of the NNBF on hydrodynamic roughness occurs on a range of scales (skin friction, form drag, etc.), and the relative contribution of each on reducing wave energy and surge propagation is poorly constrained over the complex morphology (again, at a range of scales) of NNBF. Similarly, the role of NNBF in sediment dynamics by affecting wave and current velocity profiles, turbulence, shear stresses, and resulting circulation patterns (and thus sediment resuspension, transport, and deposition) is poorly understood. Gradients in such processes due to the emplacement of NNBF can cause morphologic change that can either be beneficial, such as causing the shoreline to prograde, thus reducing flooding, or detrimental, such as erosion that undermines the NNBF or causes coastal erosion that increases flooding.

With regard to influence on ecological processes, opportunities for improving modeling of NNBF suitability include understanding the role of submerged aquatic vegetation in wave attenuation, better characterizing the influence of sunlight/tree canopy in the establishment of native vegetation along the shoreline, understanding the effects of ice on the survival of onshore and coastal vegetation, and predicting the shift in native species' locations as a result of changing climatic patterns.

Modeling in support of NNBF implementation needs to occur at three scales: (1) structural, (2) geometric, and (3) geographic scale. Modeling at structural scale (order <10s of cm) is necessary to maximize success relative to the local environmental conditions and stressors, both climate and land based. Modeling at geometric scale (10s cm to 10s of m) is needed to optimize restoration or installation design to maximize wave energy attenuation and ecological management objectives. Lastly, modeling at the geographic scale (>10s of m) is fundamental to optimizing the location of the NNBF to maximize coastal flooding hazard risk reduction and ecological success, such as making the site self-supporting via larval transport. Additional considerations for site-suitability modeling (including sociopolitical modeling) can help support successful NNBF implementation and performance (Balasubramanyam and Howard 2019)

Quantifying other benefits

NNBFs can provide both flood risk reduction and a suite of co-benefits that are valued by society, such as improved aesthetics, habitat enhancement, cleaner water, increased food production, and more commercial, educational, and recreational opportunities through improved access (Barbier *et al.* 2011). The diversity of co-benefits is a positive for NNBFs, but it does present a challenge in policy and permitting settings that are more accustomed to assessing single-purpose projects. For example, existing benefit-cost analyses (BCAs) tend to focus only on damage reduction in dollars when determining the benefit of a proposed project, ignoring diverse co-benefits that may be difficult to account for in an economic framework.

In order to progress towards BCAs that sufficiently address co-benefits, it is necessary to first focus on analyzing NNBF designs in all their diverse forms and variations and all known, as well as potential, co-benefits they may influence. Such analyses are still nascent despite substantial conversations around estuarine and coastal ecosystem services and the "blue economy" over the past decade. Ecosystem services valuations (Blair *et al.* 2014) can provide a starting blueprint on how to conduct BCAs and can help organize the evaluation of NNBF design and operation that may be well informed but doesn't automatically or easily translate to economic assessments (Rosov *et al.* 2016; Hannides *et al.* 2019).

Even while quantifying only the flood protection benefits of NNBF, there are often assets inside the region protected by NNBF that, if flooded, affect a much broader region and the status of co-benefits. For example, flooding that results in power stations and sewage facilities going offline negatively impacts the broader region of service delivery that may extend outside of the flooded area itself. BCAs could be more effective and useful when assessing the social and economic impact of these greater extent of influence.

Moreover, even when there are existing methods to qualitatively and quantitatively assess the risk reduction and co-benefits of NNBF, there are associated challenges, including the lack of consistent and user-friendly methodology and lack of institutionalized project processes requiring this type of assessment. Support for NNBF to increase coastal resilience could be allocated from federal pre-disaster hazard mitigation, coastal storm risk reduction projects, and disaster recovery funding (e.g. funding allocated for recovery from Superstorm Sandy or the 2017 and 2018 hurricane seasons). Defining coastal resilience more holistically (i.e. to include NNBF co-benefits) may be a potential avenue to transform existing methodologies and assessment procedures to incorporate co-benefits. In addition, support for NNBF conservation and restoration would greatly improve coastal resilience, e.g. the Mesoamerican

Reef (Reguero *et al.* 2019), or through new resilience insurance mechanisms for wetland restoration projects (Reguero *et al.* 2020).

STAKEHOLDER ENGAGEMENT AND COMMUNICATION

Engagement within communities and across multi-disciplinary teams is important because of the diversity of individuals and groups involved with NNBF decisions, as well as the increased likelihood of multiple benefits and, therefore, multiple beneficiaries, of NNBF projects. For example, NNBF can provide social and ecological benefits in addition to risk reduction. Research has overwhelmingly concluded nature provides immense benefits to the overall well-being of people, and although research on the socioeconomic benefits from nature-based approaches is still in the early stages, preliminary results are encouraging, particularly in more urban communities (Elmqvist *et al.* 2015; Keeler *et al.* 2019). Therefore, NNBF projects present an opportunity to bring together social and natural scientists, engineers, landscape architects, tribes and indigenous stewards of coastal lands, as well as other stakeholders, to develop projects with optimized benefits. This opportunity extends to methods of stakeholder engagement. NNBF projects can benefit from community/science interaction and point to a more community-organized resilience effort than a more specialized, steel or concrete solution. Guidance on stakeholder engagement for NNBF projects is available in Chapter 3 of the international NNBF guidelines (Bridges *et al.* 2021).

Stakeholder access to and proper application of technical information for the design and implementation of NNBF projects for coastal risk reduction is limited. To partially address this limitation, the U.S. Coral Reef Task Force is developing a guidance document on coral reef restoration proposals for federal hazard mitigation funding. This effort will result in a "How-To Guide" covering a range of project application development elements including: project scoping, identification of the project team, selection of site(s), assessment of alternatives, benefit cost analysis, identification of regulatory requirements, and potential funding opportunities. Further, Blair *et al.* (2014) highlighted the role agricultural or Sea Grant extensions can play in bringing together experts to valuate ecosystem services affected by restoration projects.

Policy

Based on new insights, federal, state, territorial, tribal, and local governments are starting to consider NNBFs as national infrastructure for their storm protection benefits, as well as other co-benefits and ecosystem services. NNBF hazard risk reduction analyses and proposed benefit-cost analyses open new policy instruments that could account for NNBF health and status. For example, dynamic coastal setbacks could define the contribution to coastal risk from NNBF degradation, similar to shoreline management for sea level rise erosion in South Carolina, Hawaii, or the United Kingdom (Harris *et al.* 2009; USACE 2018; Williams *et al.* 2017).

From a state, territorial, and local government perspective, prioritization of NNBF is variable, ranging from being recommended to being required by law (Hilke *et al.* 2020). For example, some states have laws requiring use of non-structural/living shoreline approaches for shoreline stabilization, unless they are proven not suitable or are located in an area where hard structures are allowed (H.B. 973). Other states promote the use of NNBFs through inclusion in statewide activity approvals, which expedites permitting for NNBFs (DNREC 2015). In addition, some states offer grants or cost-shares to support implementation (WDFW n.d.). In a similar fashion, some local governments encourage the use of NNBFs or softer approaches through identification of shoreline reaches where hard structures are not allowed (Town of East Hampton 2019). As more data are collected to document the performance of NNBFs, along with greater outreach and stakeholder engagement, support for policies prioritizing NNBFs may grow. Post-implementation of NNBFs, zoning ordinances, and local policies should protect the interests of local residents to avoid unintended consequences by virtue of improved aesthetics and increased land values.

Standardization

There is a need to standardize both the terminology associated with NNBFs, as well as standardized frameworks for assessing the variety of benefits NNBFs can offer. Terminology associated with coastal NNBF is diverse, with terms like "Living Shorelines" (NOAA n.d.) and "Soft Shore Solutions" (Griffiths 2019) used sometimes to refer generally to NNBFs, and sometimes to describe specific categories of NNBFs. A lack of consistency can hinder national and international conversations and delay the adoption of best practices.

Access to and standardization of guidance and region-specific engineering standards would also help to better allocate funding to preferred projects or improve permitting of NNBFs. Specifically:

1. **"Standardization" of NNBF terminology** has been attempted within the international NNBF guidelines (Bridges *et al.* 2021) to deliver *consistency of terminology*. There is a specific glossary of terms included.

2. **Guidance/engineering standards.** The international NNBF guidelines (Bridges *et al.* 2021) help standardize terminology. They provide a consistent framework, principles, and steps to follow. However, it was not appropriate to deliver a detailed set of algorithms for calculating the flood or coastal erosion risk reduction performance of NNBFs, because (a) in many cases this would duplicate what is already available in other more general coastal engineering guidance and (b) the level of maturity of the science is highly variable between different types of NNBFs. In regard to (b), we can note, for example, mature information is available on the wave attenuation performance of coral reefs and on the attenuation and sediment process mechanisms associated with beach/dune systems. On the other hand, information on the effectiveness of SAV is still relatively immature

3. **Assessment of regional approaches.** There are regional differences that might be driven by the following:

a. *Culture* — the extent to which there is openness amongst coastal managers and communities to the introduction of new approaches

b. *Governance* — different governance arrangements may be influential (e.g. the extent to which hard structures are allowed on the exposed shoreline); and

c. *Geography/habitat related* — some habitats, such as mangroves or coral reefs, cannot exist in specific weather climates or environments, such as open-coasts versus estuaries.

At the same time, given the site-specific nature of NNBFs approaches and the wealth of local knowledge pertaining to natural coastal systems, standardization may have the unintended consequence of excluding the sheer variability in coastal habitats across the world. As such, we recommend a broad framework for characterizing NNBFs, allowing for local-level differentiation in the way these approaches are implemented.

NEEDS AND RECOMMENDATIONS FOR COASTAL ECOSYSTEMS

Recommendations for scientific research, observations, community engagement, and policies aim to help prioritize future research and investments. The USCRP is specifically interested in how a better understanding of coastal physical processes interactions with resilient coastal ecosystems (e.g. climate change, depositional environment, hydrodynamics) may benefit the following.

Observations and research

• Quantify the resulting hazard risk reduction of existing and potential NNBFs.

• Improve forecasting by integrating frequent and new data points into existing and novel models.

• Collect environmental data to calibrate and validate models to predict future impacts on coastal ecosystems and their evolution due to anthropogenic stressors (land-based pollution, overfishing, coastal development), climate change, and sea level rise.

• Improve understanding of the ecological performance and adaptive capacity of coastal ecosystems.

• Develop strategies to evaluate physiological and behavioral responses of biological organisms and systems to naturally varying environments.

Infrastructure, policies, and outreach

• Build on successful models of science-based regulations (e.g. federal catch limits).

• Make data publicly available when possible and promote discoverability through centralized clearinghouses.

• Empower users through co-management (e.g. capitalize on private-sector vessels as platforms for data collection) and develop adaptive co-management approaches (e.g. comprehensive ecosystem-based management).

• Standardize ecosystem and nature-based feature terminology, taking into consideration the regional and local perspectives.

• Standardize benefit assessment frameworks to include non-monetary benefits and allow both private sector (e.g. insurance, green bonds, special purpose districts) and public sector (e.g. pre-disaster mitigation funds, post-disaster restoration funds, green climate funds) funds be used for ecosystem recovery and restoration.

SUMMARY OF RECOMMENDATIONS

Coastal systems are complex, vital to our national economy, and the conditions influencing them are highly connected. The interactions between pollutants and resilient ecosystems are a key component of the interconnected coastal environment; however, much is unknown about these exchanges. In many instances, useful data are available for these systems and related environmental metrics; however, the combined influences and feedbacks among the coastal environment, climate, and human modifications require additional research and environmental data collection. Coastal conditions tend to be highly variable on short temporal and spatial scales. For example, wave patterns, tidal flushing, and day/night respiration alters environmental conditions including velocities, temperature, DO, pH, and salinity. Most studies have focused on static conditions to evaluate the response of dynamic coastal systems, thus the impact of naturally varying environmental conditions is an important area warranting further study (Boyd *et al.* 2016). This paper highlights the USCRP's specific interest in how a better understanding of coastal physical processes interactions with coastal pollutants and resilient ecosystems (e.g. climate change, depositional environment, hydrodynamics, climate change) may benefit the following.

Lower cost and rapid response tools are needed to help managers respond to pollutant threats. Early warnings, event response thresholds, and regulations based on science will continue to protect wildlife and human health. Forecasting can be improved by integrating data from these tools into existing and new models. Models will also be needed to help us understand the scale and direction of anticipated changes in these systems. For example, how will increasing storm frequency and sea-level rise in combination shape future coastal habitats?

To develop appropriate mitigation strategies for pollutants and resilient ecosystems, an assessment of potential threats to human and ecosystem health, as well as benefits that may be accrued by reducing these risks, is needed. For example, what are the benefits to society at-large accrued from reducing pollutants, and what is the adaptive capacity of coastal ecosystems?

The development and implementation of co-management approaches, publicly discoverable data and tools, and education will empower users to become better stewards of these systems.

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