EXHIBIT F

IN THE UNITED STATES DISTRICT COURT
FOR THE EASTERN DISTRICT OF NORTH CAROLINA
SOUTHERN DIVISION
No. 4:20-CV-151-FL

NORTH CAROLINA COASTAL FISHERIES REFORM GROUP, JOSEPH WILLIAM ALBEA, DAVID ANTHONY SAMMONS, CAPTAIN SETH VERNON, CAPTAIN RICHARD ANDREWS, and DWAYNE BEVELL, Plaintiffs, v. CAPT. GASTON LLC; ESTHER JOY, INC., HOBO SEAFOOD, INC.; LADY SAMAIRA, INC.; TRAWLER CAPT. ALFRED, INC.; TRAWLER CHRISTINA ANN, INC.; TRAWLERS GARLAND and JEFF, INC.; and NORTH CAROLINA DEPARTMENT OF ENVIRONMENTAL QUALITY, DIVISION OF MARINE FISHERIES, Defendants.	**AFFIDAVIT OF DAVID ANTHONY SAMMONS ON BEHALF OF PLAINTIFF NORTH CAROLINA COASTAL FISHERIES REFORM GROUP**

I, David Anthony Sammons, first being duly sworn, depose and say:

1. I am over the age of 18, have no disability, have personal knowledge of, and am competent to testify as to the matters set forth herein.
2. I am a Member of the North Carolina Coastal Fisheries Reform Group ("NCCFRG"), and currently serve as its Chief Financial Officer.
3. NCCFRG is a nonprofit membership organization that has long worked to protect North Carolina's coastal and marine public trust resources by engaging with stakeholders, the

State of North Carolina, and the public to promote a wide range of sustainable fisheries practices. This range of fisheries practices includes, but is not limited to, the use of certain types of nets in commercial fishing operations, catch limits, fishing licensing reform, greater reporting and oversight of bycatch hauls, and reduction of bycatch disposal into North Carolina coastal waters.

4. NCCFRG advocates for sustainable fisheries practices to restore the coastal environment and fisheries stocks for current users, including its members, and for future generations.

5. NCCFRG's membership includes recreational fishermen, conservationists, outdoorsmen, and small business owners who rely on North Carolina coastal waters and their fisheries for their livelihoods and personal recreation and enjoyment. NCCFRG's members regularly visit North Carolina coastal waters to fish, boat, and otherwise recreate.

6. NCCFRG's members who are recreational fishermen regularly fish North Carolina coastal waters for the same species of finfish that are caught as bycatch by shrimp trawling nets, injured, killed, and disposed of in large volumes by the Defendant Trawling Companies.

7. Defendant Trawling Companies' operations drag trawl nets along the bottom of North Carolina coastal waters, which damages natural habitats and disturbs sediments that become re-suspended in the water. Exhibit A at 13-20. Commercial shrimp trawlers, including Defendant Trawling Companies, create massive, miles-long sediment plumes in North Carolina coastal waters. These expansive sediment plumes are visible in satellite imagery. A representative sample of such satellite imagery captured on October 14, 2020 is attached as Exhibit B.

8. Suspended and re-suspended sediment harms finfish, for example by clogging their gills and thereby diminishing their ability to breathe. Exhibit C at 341-42.

9. Suspended and re-suspended sediments also block sunlight from reaching submerged aquatic vegetation that needs regular sunlight to survive. Exhibit D.

10. The bodies of finfish and aquatic vegetation that ultimately succumb to sediment pollution then add additional organic matter to the water. In large quantities, organic matter leads to the depletion of dissolved oxygen in a water body, which directly harms finfish and other aquatic species. *See* Exhibit E.

11. Sediments also contain nutrients that are re-suspended and re-dispersed into the water when disturbed by commercial shrimp trawling activities. Exhibit F.

12. It is well-known that nutrient pollution in water bodies can lead to over-growth of aquatic vegetation, phytoplankton, and algae, including cyanobacteria (blue-green algae) that is toxic to humans and animals. Exhibit G at 84-101. This overgrowth depletes available dissolved oxygen in the water body, which in turn harms or even kills finfish and other aquatic species. Exhibit E.

13. North Carolina's coastal waters, including the Albemarle-Pamlico Sound, already suffer from regular, documented algal blooms. Exhibit H. NCCFRG's members, as regular users of North Carolina's coastal waters, stand an increased risk of exposure to these algal blooms, as well as increased likelihood that these algal blooms disrupt their economic and recreational opportunities.

14. Suspended and re-suspended sediments, increased organic matter, and the resulting turbidity also diminish North Carolina coastal waters' recreational and aesthetic qualities for NCCFRG's members.

15. Defendant Trawling Companies also capture, injure, kill, and dispose of large volumes of finfish back into North Carolina's coastal waters. This disposal degrades the aquatic

environment and the ecosystem which surviving finfish rely on for survival and which NCCFRG member rely on for their livelihoods and recreation.

16. Bycatch disposal greatly increases the amount of organic matter in North Carolina coastal waters. As already mentioned, large increases of organic matter in a water body lead to decreased dissolved oxygen levels that harm or kill finfish and other aquatic species. This disposal perpetuates a cycle of an overabundance of organic matter, which in turn leads to an overburden of aquatic vegetation, phytoplankton, and algae, and subsequent dangerous decreases in dissolved oxygen.

17. To address Defendant Trawling Companies' significant, ongoing sediment dredging and bycatch disposal practices, NCCFRG assessed its existing financial, volunteer, and in-house expert resources and determined that it would need to greatly expand the organization's existing capacity.

18. Although NCCFRG has previously used its members' volunteered time and resources to conduct its work, its engagement on Defendant Trawling Companies' dredging and bycatch disposal practices, which constitute Clean Water Act violations traceable to the defendants' actions, has required it to hire private counsel, engage in greatly expanded fundraising efforts, secure the commitment of additional in-house volunteer hours, divert volunteer efforts and available funding from other NCCFRG sustainable fisheries initiatives, and otherwise expand its base of subject area experts.

19. Pursuing Defendant Trawling Companies' Clean Water Act violations is consistent with NCCFRG's mission to advocate for sustainable fisheries practices, but has required NCCFRG to do more than simply re-allocate existing resources.

20. This significant increase in funding, expertise, and in-house capacity has been and continues to be necessary to protect NCCFRG's and its members' concrete, particularized interests in North Carolina coastal waters and fisheries from Defendant Trawling Companies' ongoing commercial shrimp trawling activities.

Further affiant sayeth not.

This the _3rd_ day of _November_, 2020.

_____ David Anthony Sammons (Nov 3, 2020 14:50 EST)

David Anthony Sammons, on behalf of Plaintiff North Carolina Coastal Fisheries Reform Group

STATE OF NORTH CAROLINA

COUNTY OF _New Hanover__

Subscribed and sworn to before me

This the __3rd__ day of __November__, 2020.

BRIAN HOSS
Electronic Notary Public
New Hanover County
North Carolina
Commission Expires 6/26/2024

_____

Notary Public

My Commission Expires: __June 26th, 2024__

Exhibit A

NOAA Technical Memorandum NMFS-SEFSC-449

U.S. Department of Commerce

National Oceanic and Atmospheric Administration

National Marine Fisheries Service

A review of the fishing gear utilized within the Southeast Region and their potential impacts on essential fish habitat

by:

Michael C. Barnette

National Marine Fisheries Service
Southeast Regional Office
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St. Petersburg, FL 33702

February 2001

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by: Michael C. Barnette

U.S. Department of Commerce
Donald L. Evans, Secretary

National Oceanic and Atmospheric Administration
Scott B. Gudes, Acting Under Secretary for Oceans and Atmosphere

National Marine Fisheries Service
William T. Hogarth, Acting Assistant Administrator for Fisheries

February 2001

NOTICE

This report should be cited as follows:

Barnette, M.C. 2001. A review of the fishing gear utilized within the Southeast Region and their potential impacts on essential fish habitat. NOAA Technical Memorandum NMFS-SEFSC-449, 62pp.

Copies may be obtained by writing:
National Technical Information Service
5258 Port Royal Rd.
Springfield, VA 22161
1-800-553-6847 or 1-703-605-6000

ACKNOWLEDGMENT

I would like to acknowledge Jeff Rester of the Gulf States Marine Fisheries Commission, for his effort in compiling his annotated bibliography of fishing impacts, as well as for his assistance and comments during the development of this paper.

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BACKGROUND

Habitat is increasingly recognized as critical to maintaining species diversity and supporting sustainable fisheries. The 1996 reauthorization of the Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act) mandated that fishery management plans (FMPs) be amended to include the description and identification of essential fish habitat (EFH) for all managed species. The Magnuson-Stevens Act defined EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity."

The Magnuson-Stevens Act also required that adverse impacts on EFH resulting from fishing activities be identified and minimized to the extent practicable. In order to minimize adverse impacts on EFH resulting from fishery-related activities, an evaluation of the various fishing gear types employed within the jurisdictions of all Fishery Management Councils was necessary. This evaluation developed into a profoundly difficult obstacle given the paucity of readily available information on the numerous types of gear utilized within the South Atlantic, Gulf of Mexico, and Caribbean. While there have been hundreds of studies published on gear impacts worldwide, the majority of these focus on mobile gear such as dredges and trawls. Furthermore, in addition to the approved gears within the various FMPs, there are many gears utilized within state and territorial waters that also needed to be evaluated due to the extension of defined EFH into coastal and estuarine waters. However, there are few, if any, habitat impact studies that have been conducted on many of these gear types. Due to the lack of specific information and regional fishery-related impact studies, the Gulf of Mexico Fishery Management Council's Generic Amendment for Addressing Essential Fish Habitat Requirements and the Caribbean Council's Essential Fish Habitat Generic Amendment were only partially approved by NOAA Fisheries.

To help remedy these deficiencies, an annotated bibliography (Rester 2000a; 2000b) was completed which compiled a listing of papers and reports that addressed fishery-related habitat impacts. The bibliography included scientific literature, technical reports, state and federal agency reports, college theses, conference and meeting proceedings, popular articles, memoranda, and other forms of nonscientific literature, but did not include studies that pertained to the ecosystem effects of fishing. While recognizing that fishing may have many varying impacts on EFH, the bibliography focused on the physical impacts of fishing activities on habitat.

In order to determine if the approximately 600 studies included in the bibliography were relevant to the Southeast Region, criteria were developed during a December 1999 EFH Workshop attended by NMFS scientists and managers. The criteria included whether the specified gear was utilized in the Southeast Region, whether it was utilized in the same manner (similar fisheries), and whether the habitat was similar. This review recognized that in many instances numerous epifaunal and infaunal species are an integral part of benthic habitat. Therefore, studies that document impacts (i.e., reduction in biomass or species diversity) to benthic communities have been included in this review.

Studies of gear types that were not applicable to the Southeast Region such as explosives, cyanide/poisons, and beam trawls were not included. Explosives and cyanide have been prohibited by the various Fishery Management Councils due to the documented habitat damage associated with those methods. The numerous studies conducted on beam trawls were excluded due to the fact that beam trawls are not a favored gear type within the region. While a study published by ICES (1973) concluded that otter trawls and beam trawls are similar in their action on the seabed and that there is no good reason for considering possible destructive effects of beam and otter trawls separately, it was felt that there were enough studies that specifically detailed otter trawls to exclude the numerous beam trawl studies. Studies documenting habitat damage resulting from anchoring or interactions with marine vessels (e.g., groundings, propeller scarring) were not considered in this review unless the activity was directly related to harvesting methods (e.g., clam-kicking, skimmer trawling, etc.). While anchors are utilized during various commercial and recreational fishing activities, anchors are not a type of fishing gear and, thus, were not considered. Based on these criteria, habitat impacts, recovery metrics, and management recommendations were extracted from the study and included in this review.

While DeAlteris et al. (1999) stated that fishery-related impacts to EFH need to be compared to natural causes, both in magnitude and frequency of disturbance, fishing can be adjusted or eliminated to complement particular habitats, whereas natural conditions continue unabated. Depending on the intensity and frequency of fishing, its impacts may well fall within the range of natural perturbations. However, Hall (1999) pointed out that while it is important to appreciate the range of natural variation in disturbance from currents, wind, and waves so that fishing can be put into context, the fact that the natural range is large in itself provides no basis for arguing that the additional perturbation imposed by fishing is inconsequential. Marine communities and their associated habitats have adapted to natural variation. Fishing impacts may introduce a variable that is beyond the range of natural impacts, potentially resulting in dramatic alterations in habitat or species composition. For example, Posey et al. (1996) suggested that deeper burrowing fauna are not affected by severe episodic storms, though they may still be impacted by fishing. The study site was at a depth of 13m and samples were collected to a depth of 15cm below the substrate. "Deeper burrowing" was not defined, but it implies fauna living at a depth of 7 - 15cm (Jennings and Kaiser 1998) which is well within the depths disturbed by trawls and dredges (Krost and Rumohr 1990). Regardless, information from studies that include comparisons of fishery-related impacts to natural events have been included in the scope of this review.

ESSENTIAL FISH HABITAT

As defined by the Magnuson-Stevens Act, EFH includes "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." Interpretation of this definition may vary, therefore, NMFS has provided further guidance to assist with the legal interpretation of EFH: *waters* - aquatic areas and their associated physical, chemical, and biological properties that are used by fish and may include aquatic areas historically used by fish where appropriate; *substrate* - sediment, hard bottom, structures underlying the waters, and associated biological communities; *necessary* - the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem; and *spawning, breeding, feeding, or growth to maturity* - stages representing a species' full life cycle.

The degree of impact from fishing activities depends in large part to the susceptibility of particular habitats to damage. EFH varies in its vulnerability to disturbance, as well as its rate of recovery. For example, due to its simple composition, sediments (i.e., sand, mud) are impacted to a lesser degree than a complex coral reef under similar treatments. Coral reefs are composed of numerous structure forming species, many that grow vertically into the water column (e.g., sponges, stony corals, gorgonians) and create a greater surface area than sediments. The vertical profile and increased surface area of coral reefs allow gear to easily become snagged or entangled, thus providing more opportunities for habitat to be impacted from fishing as compared to sediments.

While NMFS and the Fishery Management Councils have jurisdiction only in Federal waters of the exclusive economic zone under the Magnuson-Stevens Act, estuarine and nearshore waters are critical to various life stages of many organisms; numerous managed species utilize estuaries and bays for reproduction or during juvenile development. Therefore, it is important to recognize these habitat areas as EFH (Table 1), as well as identifying potential threats to those habitats. A brief summary of the more recognizable habitat types follows. Further discussion on EFH, including geographical mapping of those habitats, may be found in the various Council EFH Amendments.

ARTIFICIAL REEFS

The National Fishing Enhancement Act of 1984 (Title II of PL 98-623) defined artificial reefs as "...a structure which is constructed or placed in waters covered under this title for the purpose of enhancing fishery resources and commercial and recreational fishing opportunities." Prior to 1985, artificial reef development projects utilized natural or scrap materials almost exclusively because of their relatively low cost and availability. With increased funding and support, many coastal states have been able to plan and execute more effective artificial reef development activities. Many programs now are taking advantage of more advanced technologies and methodologies to design materials and structures for specific fisheries management objectives.

	ESTUARINE COMPONENT	MARINE COMPONENT
GULF OF MEXICO	estuarine emergent wetlands; mangrove wetlands; SAV; Algal flats; mud, sand, shell, and rock substrates; estuarine water column	water column; vegetated bottom; non-vegetated bottom; livebottom; coral reefs; artificial reefs; geologic features; continental shelf features
SOUTH ATLANTIC	estuarine emergent wetlands; estuarine scrub and shrub mangroves; SAV; oyster reefs and shell banks; intertidal flats; palustrine emergent and forested wetlands; aquatic beds; estuarine water column	livebottom; coral and coral reefs; artificial reefs; *Sargassum*; water column
CARIBBEAN	salt marshes; mangrove wetlands; intertidal flats and salt ponds; soft bottom lagoons; mud flats; sandy beaches; rocky shores	water column; SAV; non-vegetated bottom; coral reefs; algal plains; geologic features; livebottom

TABLE 1. ESTUARINE AND MARINE EFH COMPONENTS WITHIN THE SOUTHEAST REGION.

The deployment of artificial structure on the seabed provides increased surface area for organisms to colonize and develop into a functioning reef over time. Algae and encrusting organisms cover the bare structure, similar to the ecological succession of newly exposed natural solid substrate. Finfish and invertebrate species are eventually attracted to the structure. Numerous pelagic and transient organisms also utilize the artificial reef as habitat. As these structures are designed primarily for the enhancement of fishing opportunities, fishing pressure may be focused over an artificial reef and result in subsequent impacts, such as line entanglement.

HARDBOTTOM AND CORAL REEFS

The majority of hardbottom in the Gulf of Mexico and South Atlantic consists of exposed limestone on which algae, coral, and sponges establish and accumulate. Hardbottom areas may be found throughout the Gulf of Mexico, especially along the west coast of Florida, as well as along the entire eastern seaboard to North Carolina. Many species important to commercial and recreational fisheries reside around banks, ledges, and small outcroppings colonized by sessile invertebrates such as hydroids, bryozoans, gorgonians, anthozoans, and algae that form complex benthic communities. Furthermore, many areas along the west coast of Florida are characterized by a thin sand veneer covering solid limestone. This layer of sand inhibits coral growth, but allows for sponge colonization. In some locales, sponges are quite abundant and provide the only substantial vertical habitat for many species.

Coral reefs have the highest biological diversity in the marine environment. Coral reefs, as opposed to encrusting, lower-profile hardbottom habitat, consist of a ridge limestone structure built by corals and algae. The calcium carbonate skeletons of living and dead corals are interlocked and cemented together by coralline algae. Over time, rubble and sand containing the shells of many other plants and animals become trapped between the skeletons adding to the reef mass. This three-dimensional structure provides a variety of refuge areas that attracts a plethora of marine species. While reefs cover only 0.2% of the ocean's area, they provide habitat to one-third of all marine fish species and tens of thousands of other species.

Hardbottom and coral reefs are perhaps the most sensitive habitat type within the Southeast Region, due to the abundance of encrusting and structure-forming species that produce complex and delicate habitats. Deep-water coral banks may be especially vulnerable to fishery-related impacts, as illustrated by the degradation of the Oculina Bank off eastern Florida. While shallow, high-profile coral reefs are generally well-mapped, patchy hardbottom, as well as deep-water pinnacles that occur throughout the Gulf of Mexico and South Atlantic are not well-mapped and frequently may be impacted by fishing activities.

OYSTER REEFS

Clusters of oyster shell, live oysters, and other commensal organisms form distinct oyster reef habitats. Oyster reefs tend to form wherever hard bottom occurs and sufficient current exists to transport planktonic food to the

filter-feeding oysters and to carry away sediment. Subtidal or intertidal reefs form in open bays, along the periphery of marshes, and near passes and cuts. They are particularly abundant along the side slopes of navigation channels where tidal exchange currents are dependable. The reef is three-dimensional since shells cemented together create an irregular surface that establishes a myriad of microhabitats for smaller species.

The value of oysters as filter-feeding organisms has long been recognized, however, the habitat that oyster reefs provide to resident and transient species may not be fully appreciated. The increased surface area of an oyster reef allows for greater species diversity than flat areas due to expanded habitation opportunities (Watling and Norse 1998). Reef structure formed by oysters creates vast interstitial spaces for small invertebrates and juvenile fish, analogous to a tropical coral reef. Impacts to oyster reefs, especially fishing activities that target oysters, directly reduce EFH and hamper the natural water-cleansing ability of oysters (Coen 1995). Furthermore, fishing activities adjacent to oyster reefs can have a significant impact. Fishing activities that have the ability to suspend large quantities of sediment can over-task the natural filtering ability of oysters and excess sedimentation can potentially stress or smother oysters, degrading EFH.

SEDIMENTS

Consolidated and unconsolidated sediments within the Southeast Region include a wide variety of coarse sands, shell hash, and fine silts and muds. Benthic areas comprised of sand are easily altered by natural environmental conditions such as currents and surge that constantly reshape surface features. Larger sized sediments (e.g., gravel, cobble, boulder) are more resilient to resuspension and are relatively static. In contrast, silt, mud, and clay are extremely susceptible to resuspension, and therefore usually accumulate in areas that are either infrequently impacted by natural events or are frequently renourished with sediments (Watling and Norse 1998). Therefore, fishing activities may have a greater effect on mud bottoms than on sand.

SUBMERGED AQUATIC VEGETATION

Submerged aquatic vegetation (SAV) is an assembly of rooted macrophytes generally found in shallow water where there is adequate light penetration to allow photosynthesis. Similar to terrestrial grasslands, SAV species establish physical assemblages of SAV beds or meadows. Also known as seagrasses, SAV provides food and habitat for waterfowl, fish, shellfish, and invertebrates; serves as nursery habitat for many marine species; produces oxygen in the water column as part of the photosynthetic process; filters and traps sediment that can cloud the water and bury bottom-dwelling organisms; protects shorelines from erosion by slowing down wave action; and removes excess nutrients, such as nitrogen and phosphorus, that could fuel unwanted growth of algae in the surrounding waters.

Two categories of SAV impact can be established: damage to the exposed plant, including leaf-shearing and burial, and disturbance to the underground stem, or rhizome. Individual leaf-shearing events do not represent a significant threat to SAV health, however, fishing activities that repeatedly shear leaves could result in SAV loss. It should be noted that impacts also range in severity depending on the species. Impacts on species that depend largely on sexual reproduction (e.g., *Halophila decipiens*) may be extreme, as flower and seed removal may hamper SAV establishment. Fishing activities that resuspend sediments attenuate ambient light, negatively impacting the photosynthetic processes of submerged plants. Furthermore, there is a potential for smothering by sediments precipitating out of the water column if the load is copious enough or the activity occurs frequently enough. For example, the growing tips of *Halophila spp.* are very close to the sediment and are extremely susceptible to burial. Disturbance to the rhizome generally presents a more serious threat to SAV survival than impacts to the exposed plant as SAV loss will occur. Fishing activities that impact the root structure of SAV undermine the ability of SAV beds to stabilize sediments and remove nutrients and should therefore be considered a serious impact to habitat.

WATER COLUMN

The dynamic environments of the estuarine and marine water column provide rich opportunities for migrating and residential biota to thrive. The water column can be defined by a horizontal and vertical component. Horizontally, salinity gradients strongly influence the distribution of biota. Horizonatal gradients of nutrients, decreasing seaward, affect primarily the distribution of phytoplankton and, secondarily, the organisms that depend on this primary productivity. Vertically, the water column may be stratified by salinity, oxygen content, and nutrients (SAFMC 1998a). The water column is especially important to larval transport. While the water column is relatively difficult to precisely define, it is no less important since it is the medium of transport for nutrients and migrating organisms between estuarine, inshore, and offshore waters (SAFMC 1998a).

WETLANDS

Wetlands, subject to periodic flooding or prolonged saturation, are quite diverse depending on their location. Wetland types include marshes, swamps, and other areas that link land and water. Because they can be composed of freshwater (palustrine) or saltwater (estuarine), wetlands can host numerous regional plant and animal species. Wetlands in the Southeast Region include the ubiquitous salt marsh and mangroves. These areas are closely linked with the terrestrial environment and they have adapted to the extremely diverse marine, atmospheric, and terrestrial environmental conditions that prevail. Therefore, physical impacts from fishery-related activities may not be a serious concern to these habitats as compared to more sensitive marine areas.

FISHERY-RELATED IMPACTS

All fishing has an effect on the marine environment, and therefore the associated habitat. Fishing has been identified as the most widespread human exploitative activity in the marine environment (Jennings and Kaiser 1998), as well as the major anthropogenic threat to demersal fisheries habitat on the continental shelf (Cappo et al. 1998). Fishing impacts range from the extraction of a species which skews community composition and diversity to reduction of habitat complexity through direct physical impacts of fishing gear.

The nature and magnitude of the effects of fishing activities depend heavily upon the physical and biological characteristics of a specific area in question. There are strict limitations on the degree to which probable local effects can be inferred from the studies of fishing practices conducted elsewhere (North Carolina Division of Marine Fisheries 1999). The extreme variability that occurs within marine habitats confounds the ability to easily evaluate habitat impacts on a regional basis. Obviously, observed impacts at coastal or nearshore sites should not be extrapolated to offshore fishing areas because of the major differences in water depth, sediment type, energy levels, and biological communities (Prena et al. 1999). Marine communities that have adapted to highly dynamic environmental conditions (e.g., estuaries) may not be affected as greatly as those communities that are adapted to stable environmental conditions (e.g., deep water communities). While recognizing the pitfalls that are associated with applying the results of gear impact studies from other geographical areas, due to the lack of sufficient and specific information within the Southeast Region it is necessary to review and carefully interpret all available literature in hopes of improving regional knowledge and understanding of fishery-related habitat impacts.

In addition to the environmental variability that occurs within the regions, the various types of fishing gear and how each is utilized on various habitat types affect the resulting potential impacts. For example, trawls vary in size and weight, as well as their impacts to the seabed. Additionally, the intensity of fishing activities needs to be considered. Whereas a single incident may have a negligible impact on the marine environment, the cumulative effect may be much more severe. Within intensively fished grounds, the background levels of natural disturbance may have been exceeded, leading to long-term changes in the local benthic community (Jennings and Kaiser 1998). Collie (1998) suggested that, to a large extent, it is the cumulative impact of bottom fishing, rather than the characteristics of a particular gear, that affects benthic communities. Unfortunately, a limitation

to many fishing-related impact studies is that they do not measure the long-term effects of chronic fishing disturbance. Furthermore, one of the most difficult aspects of estimating the extent of fishing impacts on habitat is the lack of high-resolution data on the distribution of fishing effort (Auster and Langton 1999).

The effects of fishing can be divided into short-term and long-term impacts. Short-term impacts (e.g., sediment resuspension) are usually directly observable and measurable while long-term impacts (e.g., effects on biodiversity) may be indirect and more difficult to quantify. Even more difficult to assess would be the cascading effects that fishery-related impacts may have on the marine environment. Additionally, various gears may indirectly impact EFH. Bycatch disposal and ghost fishing are two of the more well documented indirect impacts to EFH. While recognizing that these are serious issues that pertain to habitat, this review does not attempt to discuss these due to the secondary nature of the impacts.

The majority of existing gear impact studies focus on mobile gear such as trawls and dredges. On a regional scale, mobile gear such as trawls impact more of the benthos than any other gear. However, other fishing practices may have a more significant ecological effect in a particular area due to the nature of the habitat and fishery. Yet there are few studies that investigate other gear types, especially static gear. Rogers et al. (1998) stated that there are few accounts of the physical contact of static gear having measurable effects on benthic biota, as the area of seabed affected by each gear is almost insignificant compared to the widespread effects of mobile gear. Regardless, static gear may negatively affect EFH and, therefore, must be considered.

The exact relationship that particular impacts have on the associated community and productivity is not fully understood. While it is clear that fishing activities impact or alter EFH, the result of those impacts or the degree of habitat alteration that still allow for sustainable fishing is unknown (Dayton et al. 1995; Auster et al. 1996; Watling and Norse 1998). Hall (1994) noted that not all impacts are negative. A negative effect at one level may sometimes be viewed as a positive effect at a higher level of biological organization – particular species may be removed in small-scale disturbances yet overall community diversity at the regional scale may rise because disturbance allows more species to coexist.

REGIONAL FISHING GEAR

The Southeast Region includes numerous diverse fisheries within the jurisdictions of the South Atlantic, Gulf of Mexico, and Caribbean Fishery Management Councils. A list of allowable gears for these fisheries is included in Appendix 1. However, there are many more fisheries that exist within the state and territorial waters along the periphery of these Councils. Some of the gear types discussed in this review are utilized solely in state or territorial waters. For example, the use of hydraulic escalator dredges, crab scrapes, and clam rakes occur strictly in state waters. While there may be associated impacts with these gear types, management responsibilities fall on the individual state authorities and are outside the auspices of the Magnuson-Stevens Act. These gear types have been included in this review due to the inclusion of state and territorial waters within the defined boundaries of EFH.

For purposes of this review, the various gear types are classified as either "mobile," "static," or "other" gear and are listed in alphabetical order. Included for each gear type is a brief description, as well as potential habitat impacts, habitat recovery metrics, and potential management measures as cited in the literature. Due to the absence of information on several gear types (e.g., harpoon, slurp gun, snare), the author has included a brief representation of potential habitat impacts for those not previously evaluated, in part based on discussion with other NMFS scientists and managers during a December 1999 EFH Workshop. A summary of potential habitat impacts developed during the December EFH Workshop may be found in Table 2.

	HABITAT TYPE							
GEAR TYPE	MUD	SAND	SAV	RUBBLE	HARDBOTTOM	OTHER		REFERENCE[1]
Otter trawl	+ +	+ +	+ +	+	+ +			Berkeley et al. 1985
Roller-rigged trawl	+ +	+ +		+	+ + +			Van Dolah et al. 1987
Frame trawl	+	0	0		+			Berkely et al. 1985
Midwater trawl						0	midwater	Auster et al. 1996
Skimmer trawl	+	+	+					
Scallop dredge	+ +	+ +	+ +	+ + +	+ + +			Auster et al. 1996
Oyster dredge	+ +	+ +	+ + +	+ +		+ + +	oyster reef	Barnette 1999
Hydraulic dredge	+ + +	+ + +	+ + +	+ + +		?	oyster reef	Godcharles 1971
Handline; hook and line					+			Barnette 1999
Bottom longline	+	+			+			SAFMC 1991
Fish trap	?	?	+ +		+ +	+	algal plain	Quandt 1999
Crab trap	?	0	+					Eno et al. 1996
Lobster trap	?	0	+		+ +			Eno et al. 1996
Clam kicking	+ + +	+ + +	+ + +	+ + +				Peterson et al. 1987a
Rake	+ +	+ +	+ +	+ +		+ + +	oyster reef	Barnette 1999
Patent tongs	+ +	+ +	+ + +	+ +		+ + +	oyster reef	Barnette 1999
Bandit gear					+			Barnette 1999
Buoy gear					+			Barnette 1999
Trolling gear					+			CFMC 1999
Trot line	+	+	+					Barnette 1999
Cast net	+		+	+				De Sylva 1954
Haul seine	+	+	+			+ +	cumulative	Sadzinski et al. 1996
Hand/Beach seine			+			+		Barnette 1999
Push net			+					De Sylva 1954
Purse seine	+	+	?			0	midwater	Auster et al. 1996
Gill net	+	+	+	?	+			Carr 1988
Fyke net	+	+	+					Barnette 1999
Trammel net	+	+	+			0	estuarine	Barnette 1999
Pound net	0	0	0			0	estuarine	Barnette 1999
Butterfly net	0	0	0			0	estuarine	Barnette 1999
Spear		0			+			GMFMC 1993
Powerhead		0			0	0	pelagic	Barnette 1999
Hand harvest		0		+	+ +			Barnette 1999
Snare		0			+			Barnette 1999

Slurp gun		0		0 / +	0 / +			Barnette 1999
Bully net	0	0	0		+			Barnette 1999
Hoop net	+	+	+		+			Barnette 1999
Harpoon						0	pelagic	Barnette 1999
Hand/Dip net					+			Barnette 1999
Allowable chemical					+			Japp and Wheaton 1975
Channel net	+	+	+					
Barrier net	?	?	?	?	+			Barnette 1999
PROHIBITED GEAR								
Explosives	+ + +	+ + +	+ + +	+ + +	+ + +			Alcala and Gomez 1987
Cyanide/Bleach					+ + +			Barber and Pratt 1998

[1]For further references, consult the Annotated Bibliography on Fishing Impacts to Habitat (Rester 2000a; 2000b).

Table 2. Summary of the Potential Physical Impacts to EFH in the Gulf of Mexico, South Atlantic, and Caribbean Developed During the December 1999 NOAA Fisheries EFH Workshop (High + + +, Medium + +, Low +, Negligible 0, Unknown ?)

MOBILE GEAR

CRAB SCRAPE

A crab scrape is composed of a net bag attached to a rigid frame with short teeth (Figure 1). This gear, used exclusively in state waters, is dragged in the shallow water of bays and estuaries to catch crabs.

IMPACTS

There are no studies available that document potential damage to habitat. However, due to their design, their use in SAV would likely result in the potential uprooting of some plants, as well as leaf shearing (Barnette personal observations). However, crab scrapes are not typically employed in vegetated areas due to the amount of plant litter that would fill the net. Penetration of the benthos by the teeth would result in sediment resuspension.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

FRAME TRAWL

Roller frame trawls (Figure 2) are primarily utilized to harvest bait shrimp in the State of Florida. They consist of a frame that holds open a net and supports slotted rollers that grip the bottom and turn freely. This motion prevents the scouring and scraping impacts primarily associated with otter trawls. Because participants in the fishery usually operate in shallow water, 9.14m (30ft) or less, frame trawls are typically limited to state waters.

IMPACTS

A study by Futch and Beaumariage (1965) found that while frame trawls gathered large amounts of unattached algae and deciduous *Thalassia testudinium* leaves, no SAV with roots attached were found in the trawl catch. Trawls with larger rollers (20.3cm; 8in diameter.) reduced the amount of bycatch material, with most drags

FIGURE 1. CRAB SCRAPE (West et al. 1994)

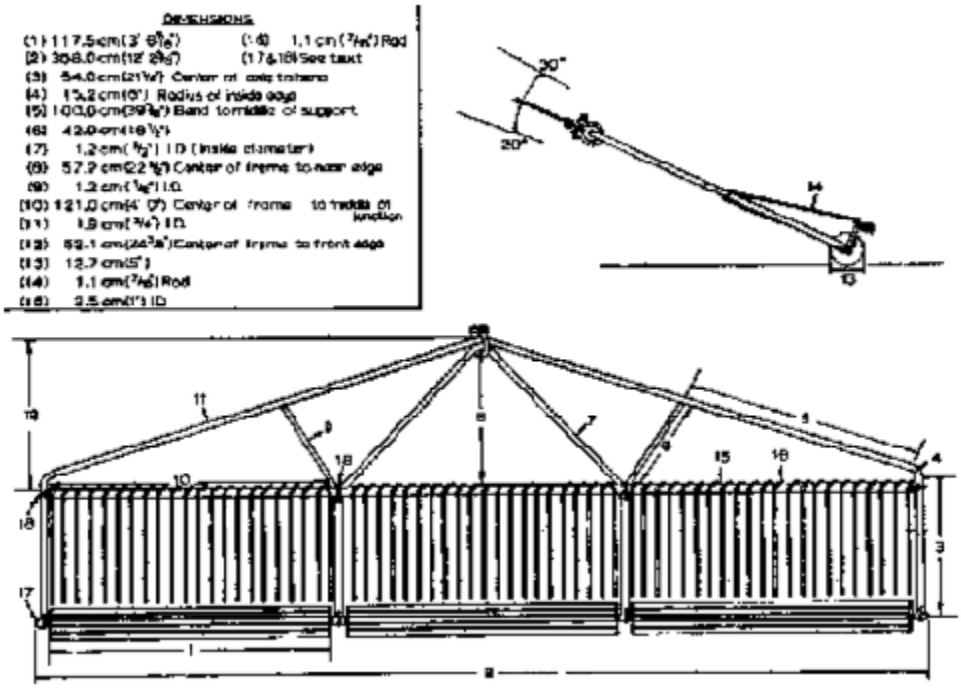


FIGURE 2. FRAME TRAWL (Tabb and Kenney 1967)

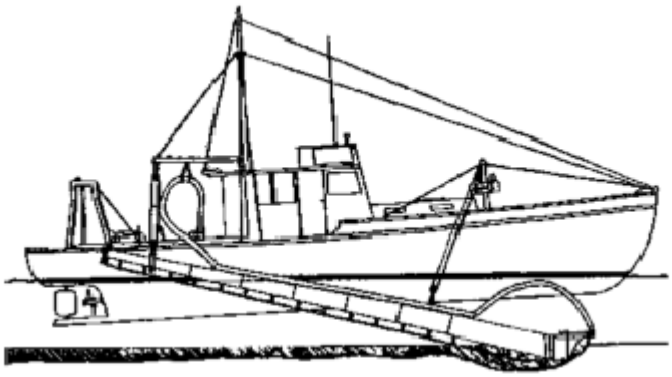


FIGURE 3. ESCALATOR DREDGE (Kyte and Chew 1975)

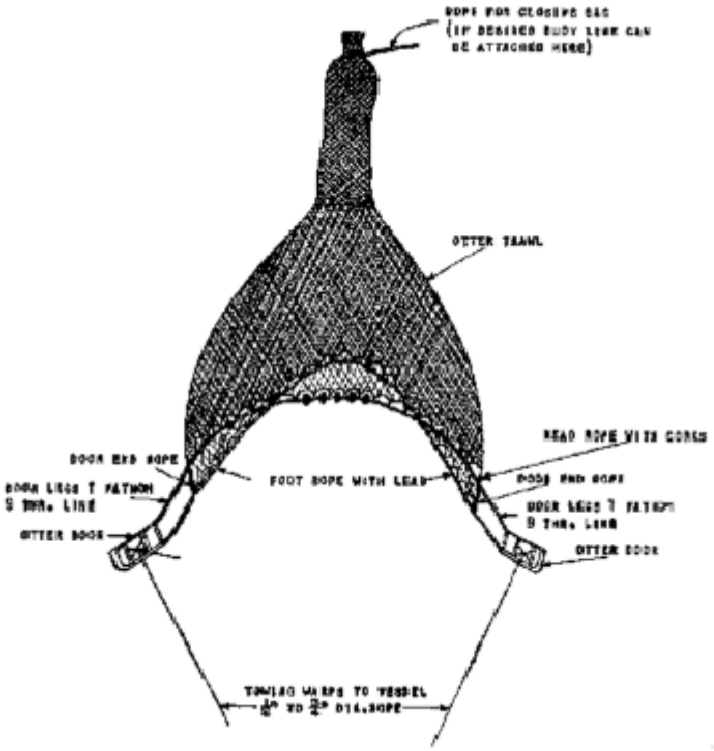


FIGURE 4. OTTER TRAWL (Richards 1955)

FIGURE 5. OYSTER DREDGE (West et al. 1994)

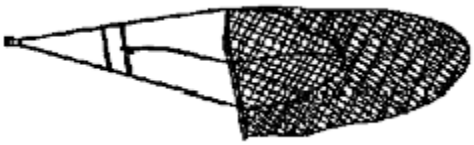


FIGURE 6. SCALLOP DREDGE (West et al. 1994)

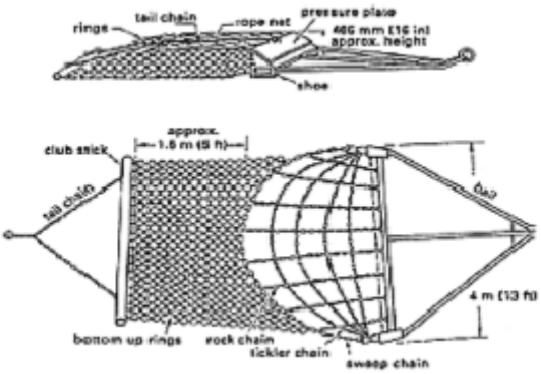


FIGURE 7. SCALLOP DREDGE (OFFSHORE)

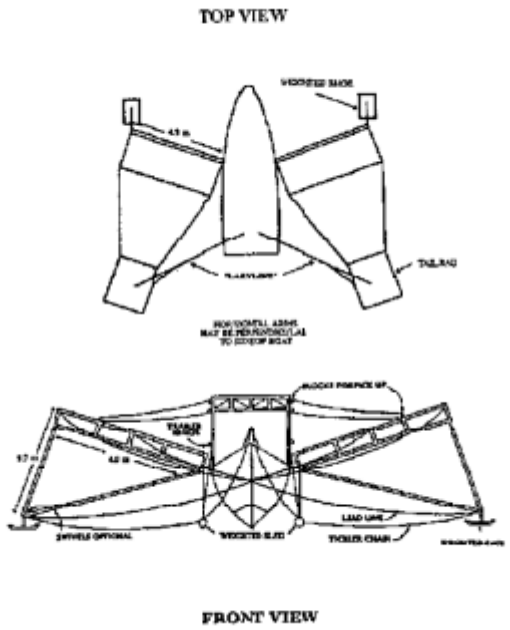


FIGURE 8. SKIMMER TRAWL (Coale et al. 1994)

collecting little or no SAV or algae. When rake teeth were extended below the rollers, they had a tendency to uproot SAV. Damage to SAV beds was noted on several occasions when the boats ran aground. The study concluded that side frame trawls do negligible damage to SAV beds. This conclusion was supported by Meyer et al. (1991; 1999), who found no significant trawl impacts on shoot density, structure, or biomass with increased trawling on turtlegrass (*Thalassia testudinium*). However, these studies did not evaluate the effects of repetitive trawling. Woodburn et al. (1957) noted that the roller on the bottom of the trawl does cause the leaves ripe for shedding to break off, though this would not negatively impact the plant itself. Higman (1952) concluded that frame trawling is not sufficient to denude vegetated areas permanently nor to damage the ecology of such locations. Additionally, Tabb and Kenny (1967), while not explicitly investigating habitat impacts, believed that roller frame trawls do no significant damage to habitat.

In contrast to studies that assessed impacts to SAV, Tilmant (1979) found a high incidence of damage to stony corals in a study that investigated frame trawl impacts to hardbottom habitat in Biscayne Bay. Frame trawls turned over or crushed 80% of *Porites porites* and *Solenastrea hyades* and damaged over 50% of sponges and 38% of gorgonians in the trawl path. Macro algae, including *Halimeda* and *Sargassum*, were heavily damaged. The primary impact on *Sargassum* was that it was torn loose from the bottom resulting in an early release to the free floating state. Tilmant (1979) found it doubtful that this action was harmful to *Sargassum* unless it occurred during early column formation. It was concluded that frame trawls have a significant impact on certain benthic organisms (Tilmant 1979). Furthermore, within dense SAV communities, removal of epibenthic algae, tunicates, sponges, and other primary producers may also be significant.

RECOVERY

Eleven months after trawling activities stopped, evidence of trawl damage on hardbottom communities was still observed but recovery was in progress (Tilmant 1979). Approximately 15% of the gorgonians encountered were previously damaged specimens which remained alive, but were lying flat on the bottom. *Porites* showed some regeneration although most *Solenastrea* encountered were dead. Algae showed complete recovery.

MANAGEMENT RECOMMENDATIONS

Futch and Beaumariage (1965) recommended that the diameter of the rollers be no less than 15.2cm (6in) and that the teeth of the rakes on the trawls should not extend below the roller. Furthermore, they recommend that boats employed in the frame trawl fishery that operate in shallow water should be of tunneled construction to prevent damage to SAV from propeller scarring. Tabb (1958) recommended that strainer bars should be rigid and aimed into the roller so that regardless of how far forward the net frame tips, the bars cannot dig into the bottom.

The results from Tilmant (1979) indicated that extensive damage occurs to hardbottom habitat from frame trawls. A logical recommendation that can be extrapolated from this study is the prohibition of frame trawling in areas where hardbottom habitat exists. Frame trawls, while causing negligible damage to SAV, are not compatible with hardbottom areas due to the damage it causes to complex vertical habitat (e.g., sponges, corals, gorgonians).

HYDRAULIC ESCALATOR DREDGE

Hydraulic escalator dredges have been utilized since the 1940s to harvest shellfish such as clams and oysters and are designed expressly for efficient commercial harvest (Coen 1995). The dredge consists of a water pump supplying a manifold with numerous water jets mounted in front of a conveyer belt that dislodges buried organisms from the sediment (Figure 3). Hydraulic escalator dredges are currently only employed in a limited shellfish fishery in South Carolina state waters.

IMPACTS

Hydraulic escalator dredges may penetrate the benthos approximately 45.7cm (18in), thus disturbance to the sediment may be substantial (Coen 1995). Increased turbidity, burial/smothering, release of contaminants, increased nutrients, and removal of infauna were offered as potential effects from dredging activities (Coen 1995). Turbidity was found to be elevated only in the immediate vicinity of the harvester operation and downcurrent of the study area to a distance of between 1.5 - 1.75km. Turbidity values returned to baseline levels within a few hours (Maier et al. 1998). Manning (1957) stated that hydraulic clam dredging can result in severe damage to oysters within a distance of 7.6m (25ft) downcurrent from the site of dredging. Enough sediment was displaced and redeposited to a distance of at least 15.2m (50ft), but not more than 22.9m (75ft) downcurrent, to cause possible damage to oyster spat. Beyond about 22.9m (75ft) there was no visible or measurable change in the experimental area. Sediment plumes caused by dredge activity were found by Ruffin (1995) to range from less than 1 to 64 hectares. Although sediment plumes increased turbidity and light attenuation at all depths, plumes in shallow water (<1.0m) caused greater increase in turbidity and light attenuation over background than did plumes in deeper waters. Plume decay is based largely on sediment size, with sand particles settling quickly while the silt/clay particles remain in suspension longer. Sites were monitored for storm disturbance to compare against dredge impacts. Storm events increased turbidity and light attenuation compared to calm days but not to the extremes obtained in sediment plumes. Storm events affect a large area at a low intensity while dredging intensely affects a more localized area. SAV subjected to decreased light penetration will inhibit reproduction, reduce propagule abundance, and structurally weaken SAV due to the need of plants growing higher into the water column (Ruffin 1995). Ruffin (1995) concluded that clam dredging increased light attenuation to the point of inhibiting SAV growth. As may be expected, hydraulic clam dredges are highly destructive to SAV within the immediate area of intensive dredging (Manning 1957; Godcharles 1971). Due to the capability of the water jets to penetrate the substrate to a depth of 45.7cm (18in), virtually all attached vegetation in its path is uprooted (Godcharles 1971). As the use of this gear is limited to a fishery in South Carolina where SAV does not exist, discussion of SAV impacts are included only to provide information on potential impacts should this gear type be considered in the future for other geographic areas where SAV may be found. Although there may be physical impacts associated with escalator dredge activity, the chemical effects apparently are not as dramatic. Dissolved oxygen, pH, and dissolved hydrogen sulfide were measured throughout the harvesting process at varying distances. No consistent patterns of depression or release were noted. Only in the direct plume of the harvester did they measure even a temporary reduction in dissolved oxygen and pH (Coen 1995). While it is recognized

that there is infaunal and epifaunal species mortality associated with escalator dredge activity, based on all evidence, these community impacts appear to be short-term (Godcharles 1971, Peterson et al. 1987a, Coen 1995). Coen (1995) noted that the escalator possibly provides a tilling effect of the bottom that has been observed to be beneficial to subtidal oyster and clam populations. Typically, shellfish dredging operations have typically not been considered to have deleterious results, since its effects are perceived to be negligible compared to natural environmental variation (Godwin 1973). Coen (1995) concluded that based on all direct and indirect evidence, the short-term effects of subtidal escalator harvesters are minimal, with no long-term chronic effects, even under worst case scenarios. Observed effects were often indistinguishable from ambient levels or natural variability.

RECOVERY

Recovery of the benthos may vary greatly depending on sediment composition. Shallower trenches with shorter residency times are typical of coarse sediments (i.e., sand), whereas trenches generated in muddy, finer sediments are typically deeper, often persisting for greater than 18 months (Coen 1995). Godcharles (1971) observed that trenches had filled in between 1 to 10 months, depending on bottom type. In regard to SAV, no trace of *Thalassia testudinium* recovery was evident after more than 1 year, though *Caulerpa prolifera* began to be establish itself in dredge areas within 86 days (Godcharles 1971).

MANAGEMENT RECOMMENDATIONS

While no management recommendations were explicitly included in any of the literature, the evidence and results provided may support the prohibition of hydraulic escalator dredge operation within SAV habitat.

OTTER TRAWL

Perhaps the most widely recognized and criticized type of gear employed in the Southeast Region is the otter trawl (Figure 4). Utilized in both state and Federal waters of the Gulf of Mexico and South Atlantic, otter trawls pursue invertebrate species such as shrimp and calico scallops, as well as finfish species such as flounder and butterfish. As the most extensively utilized towed bottom-fishing gear (Watling and Norse 1998), trawls have been identified as the most wide-spread form of disturbance to marine systems below depths affected by storms (Watling and Norse 1998; Friedlander et al. 1999).

IMPACTS

The otter trawl is one of the most studied gear types, thus, there is a wealth of information on its potential impacts to habitat. Jones (1992) broadly classified the way a trawl can affect the seabed as: scraping and ploughing; sediment resuspension; and physical habitat destruction, and removal or scattering of non-target benthos. The following discussion attempts to group documented impacts into either physical-chemical (e.g., sediment resuspension, water quality) or biological impact categories. In many instances documented habitat impacts overlap these categories.

Physical-Chemical Repercussions

The degree to which bottom trawls disturb the sediment surface depends on the sediment type and the relationship between gear type, gear weight, and trawling speed (ICES 1991). Various parts of trawl gear may impact the bottom including the doors, tickler chains, footropes, rollers, trawl shoes, and the belly of the net. While the components of trawl gear are similar, trawl design may vary greatly. Potential impacts may be shared by all otter trawls, but differences in the weight of trawl doors, footrope design, and operation (tow times), will result in a broad spectrum of impact severity. Furthermore, the number and weight of tickler chains vary the degree of disturbance: Margetts and Bridger (1971) concluded that the cumulative effect of tickler chains is likely to emulsify the sediment to a depth proportional to the number of chains. Additionally, the cumulative effect of intense otter trawling is as important as gear weight and design in impacting the benthos (Ball et al. 2000). Although the effect of one passage of a fishing (trawl) net may be relatively minor, the cumulative effect and intensity of trawling may generate long-term changes in benthic communities (Collie et al. 1997).

Trawl gear disturbs the benthos as it is dragged along the bottom. Otter trawl doors, mounted ahead and on each side of the net, spread the mouth of the net laterally across the sea floor. The spreading action of the doors results from the angle at which they are mounted, which creates hydrodynamic forces to push them apart and, in concert with the trawl door's weight, also to push them toward the sea bed (Carr and Milliken 1998). The doors, due to their design and function, are responsible for a large proportion of the potential damage inflicted by a trawl. The footrope runs along the bottom of the net mouth and may be lined with lead weight and rollers. On relatively flat bottom, it is expected that the footrope would not have a major effect on the seabed and its fauna (ICES 1995). However, in areas of complex benthic habitat the footrope would likely have more impact with the benthos. The South Atlantic Calico Scallop FMP noted that during the early years of the calico scallop fishery, large quantities of benthic material was removed by trawlers. Reports were received during numerous meetings about entire "rocks" being removed. One individual provided a print-out from a depth sounder which indicated a large amount of bottom relief in a particular area prior to the calico scallop fishery. Similar bottom plots after the calico scallop fishery operated in that area indicated a relatively flat bottom (SAFMC 1998b). Additionally, while the footrope generally causes little physical substrate alteration aside from smoothing of bedforms and minor compression on relatively flat bottoms (Brylinsky et al. 1994), these minor compressions can lead to sediment "packing" after repeated trawling activity on the same general areas (Schwinghammer et al. 1996; Lindeboom and de Groot 1998). Further compression can result from the dragging of a loaded net (cod end) along the bottom. The remaining path of the trawl is influenced by the ground warps which, while not in direct contact with the seabed, can create turbulence that resuspends sediment (Prena et al. 1999).

Trawl gear, particularly the trawl doors, penetrates the upper layer of the sediments which liquefies the affected sedimentary layers and suspends sediment in the overlying water column. This sediment "cloud" generated by the interaction of the trawl gear with the benthos and the turbulence created in its wake contributes to fish capture (Main and Sangster 1979; 1981). The appearance of the sediment cloud, but not its size, is governed by the type of seabed. Brief observations on different seabed types show that soft, light-colored mud produces the most opaque and reflective type of cloud and the fine mud remains in suspension much longer than coarse sand. Studies of sediment disturbance by trawls vary greatly, though it can be concluded that benthic habitat areas composed of fine sediments (e.g., clay, mud) are affected to a greater degree than those with coarse sediments (e.g., sand). In sandy sediments, otter boards cannot penetrate deeply due to the mechanical resistance of the sediment, and the seabed in sandy areas is more rapidly restored by waves and currents (DeAlteris et al. 1999). Short-term alterations to sediment size distribution result from the various rates of redeposition of suspended sediments; as noted before, coarse grains (i.e., sand) settle out rapidly while fine grains (i.e., silt) settle out relatively slowly. In general, resuspended sediments settle out of the water column at a rate inversely proportional to sediment size (Margetts and Bridger 1971). Transport of fine grained sediments away from trawled areas due to this slow settling period may result in permanent changes to the sediment grain size of a trawled area. Again, this effect will be more pronounced in mud/silt habitats than in habitat areas consisting of heavier sand. For example, suspended sediment concentrations of 100-500mgl^{-1} were recorded 100m astern of shrimp trawls in Corpus Christi Bay, Texas (Schubel et al. 1979), an estuary dominated by muddy sediments. The same study estimated that the total amount of sediment disturbed annually as a result of shrimp trawling was 25-209,000,000m^3, which is 10-100 times greater than the amount dredged during the same period for maintenance of shipping channels in the same area.

ICES (1973) concluded that the physical effects of trawling in tidal waters can not be permanent. However, it is possible that frequently repeated trawling of one ground with a mixed sediment type bottom in strongly tidal waters might ultimately alter the nature of the bottom towards being predominantly coarse sand because the finer particles are carried away to settle elsewhere. In deeper waters, impacts may be more profound and longer lasting. Engel and Kvitek (1998) investigated two adjacent areas in 180m of water to determine the differences between a heavily trawled site and a lightly trawled site. The data indicated that intensive trawling significantly decreased habitat heterogeneity. Rocks and mounds were less common and sediments and shell fragments were more common in the highly trawled area. Rocks and mounds were more abundant in the lightly trawled area, as well as the amount of flocculent matter and detritus. They theorized that less trawling most likely results in an area with more topographical relief and allows for the accumulation of debris, whereas consistent trawling

removes rocks, smooths over mounds, and resuspends and removes debris. Likewise, Kenchington (1995) found that sand ripples were flattened and stones were displaced after a trawl passage. Churchill (1989) modeled sediment resuspension by trawling and found that this may be a primary source of suspended sediment over the outer shelf where storm-related bottom stresses are weak.

Otter trawl doors were found to have a maximum cutting depth of 50 - 300mm (Drew and Larsen 1994) and, according to Schubel et al. (1979), the footropes of shrimp trawlers in Texas disturbed approximately the upper 50mm of the sediment. Schwinghamer et al. (1996) observed that while the trawl doors may leave scours or depressions, the overall surface roughness is reduced by trawling activity. Ripples, detrital aggregations, and surface traces of bioturbation are smoothed over by the mechanical action of the trawl and the suspension and subsequent deposition of the surface sediment. In general, the passage of an otter trawl was found to have a minor physical and visual impact on the soft sedimentary seabed, represented by a flattening of the normally mounded sediment surface and some disturbance of the sessile epifauna (Lindeboom and de Groot 1998). The potential to suspend sediments varies greatly, in large part due to the type of sediment a trawl is working on. Regardless, the suspension of sediments, whether fine silt or coarse sand, impacts the chemical and physical attributes of water quality.

The resuspension of sediments may influence the uptake or release of contaminants and, depending on the frequency of disturbance, the nature of the contaminant(s). Clearly, such effects may be more significant where contaminant burdens are relatively high, e.g., near areas affected by major industrialization (ICES 1995). Repetitive trawling on the same ground may enhance nutrient release from sediments and that estimates of average trawling effort for large areas may be unsuitable for estimating these effects (ICES 1995). This has important implications on nutrient cycling in areas that are regularly trawled. Pilskaln et al. (1998) found that impacts include burial of fresh organic matter and exposure of anaerobic sediments; large nutrient delivery to the water column, possibly impacting primary production; increase in nitrate flux out of the sediments; and reduced dentrification (conversion of remineralized nitrogen into N_2 gas). All of these may have desirable or undesirable ecosystem impacts. An increase in nitrate fluxes to the water column may alter primary production (phytoplankton), potentially benefitting fisheries, or stimulating deleterious phytoplankton growth that results in harmful algal blooms (Pilskaln et al. 1998).

Increased water turbidity as a result of trawling activity has the potential to compress the width of the euphotic zone, wherein light levels are sufficient to support photosynthesis (North Carolina Division of Marine Fisheries 1999). The magnitude of this effect depends on sediment size, duration and periodicity of the trawling event, gear type, season, and site-specific hydrographic and bathymetric features (Paine 1979; Kinnish 1992). Dredging studies would indicate that the effect of turbidity is greatly dependent on local conditions. Windom (1975) found that sediment resuspension caused by dredging operations significantly reduced phytoplankton growth in a naturally clear estuary (South Florida) but not in a naturally turbid estuary (Chesapeake Bay). Additionally, increased turbidity resulting from trawling activities may reduce primary production of benthic microalgae. This may have serious consequences as benthic microalgae support a variety of consumers and can be a significant portion of total primary production (Cahoon and Cooke 1992; Cahoon and Tronzo 1992; Cahoon et al. 1990; 1993). Increased turbidity also has may reduce the foraging success of visual predators (Minello et al. 1987) and contribute to the mortality of organisms by impeding the normal functioning of feeding and respiratory structures (Sherk et al. 1975).

Sediment resuspension may increase the amount of organic matter resulting from enhanced primary production and may stimulate heterotrophic microbial production. If the amount of resuspended organic material is copious, sustained proliferation of heterotrophic microflora will reduce the dissolved oxygen content within the water, and widespread hypoxia or anoxia could ensue to the detriment of benthic and pelagic fauna (West et al. 1994). Conversely, oxygen penetration into the sediment might be enhanced through trawling activity, resulting in shifts in mineralization patterns and redox-dependent chemical processes. Among other consequences, a change from anaerobic to aerobic conditions facilitates the degradation of hydrocarbons.

As Kaiser (2000) pointed out, bottom trawls are designed to stay in close contact with the seabed and an inevitable consequence of their design is the penetration and resuspension of the seabed to some extent. While it is possible to reduce the direct physical forces exerted on the seabed by modifying fishing practices, the benefits are questionable and catches would most certainly suffer. Despite attempts to improve gear design, as long as bottom dwelling species are harvested using towed gear, there will be inevitable sediment resuspension.

Biological Repercussions

The physical disturbance of sediment, such as the ones previously discussed, can also result in a loss of biological organization and reduce species richness (Hall 1994). In general, the heavier the gear and the deeper its penetration of the sediment, the greater the damage to the fauna. Impacts also will vary depending on type of habitat the gear is working. Gibbs et al. (1980) determined that shrimp trawling occurring within a sandy estuary had no detectable effect on the macrobenthos. After repeated trawls the sea bottom appeared only slightly marked by the trawl's passage. However, Eleuterius (1987) noted that scarring due to shrimp trawls in Mississippi SAV was common, especially in deeper water. Trawling activities left tracks and ripped up the margins of the beds, and great masses of seagrass were often observed floating on the surface following the opening of shrimp season. Furthermore, Wenner (1983) noted that the use of an otter trawl on hardbottom habitat may inflict considerable damage. The net damages the sponge-coral habitat by shearing off sponges, soft corals, bryozoans, and other attached invertebrates. Therefore, it is not necessarily that trawl gear is doing a constant level of damage, but rather particular habitats are more vulnerable to impacts than others.

Numerous studies cite specific, direct biological impacts to habitat such as the reduction of algal and SAV biomass (Tabb 1958; Fonseca et al. 1984; Bargmann et al. 1985; Peterson et al. 1987a; Sánchez-Lizaso et al. 1990; Guillén et al. 1994; Ardizzone et al. 2000). Gelatinous zooplankton and jellyfish, which provide habitat to juvenile and other fish species, are greatly impacted as they pass through the mesh of mobile gear (Auster and Langton 1999). Fishing activity may reduce the size and number of zooplankton aggregations and disperse associated fishes. Furthermore, there is a directed trawl fishery for cannonball jellyfish in the Gulf of Mexico. While this fishery removes jellyfish which may provide habitat for juvenile fish, otter trawls utilized in this fishery do not interact with the benthos. Trawls in the Gulf of Mexico and South Atlantic have been noted to impact coral habitat, damaging and destroying various colonies (Moore and Bullis 1960; Gomez et al. 1987; Bohnsack personal observation). Loss of sponges and associated cnidarian benthos has been documented to lead to a reduction in fish catch (Sainsbury 1988; Hutchings 1990). Sponges are particularly sensitive to disturbance because they recruit aperiodically and are slow growing in deeper waters (Auster and Langton 1999). Bradstock and Gordon (1983) observed that trawling virtually destroyed large areas dominated by encrusting coralline growths (bryozoans), reducing colony size and density. Probert et al. (1997) documented the bycatch of benthic species that occurs in a deep-water trawl fishery and noted the vulnerability of pinnacle communities and deep-water coral banks such as the *Oculina* habitat area of eastern Florida. Van Dolah et al. (1983; 1987) conducted experimental trawl surveys over hardbottom habitat consisting of coral and sponge off the coast of Georgia. A single pass of an otter trawl on this habitat damaged all counted species (Van Dolah et al. 1983; 1987). However, only the density of barrel sponges was significantly decreased by trawling activities. It should be noted that these studies did not investigate the cumulative impacts of trawls. The repetitive effects of trawling over the same area can be expected to have more severe consequences to benthic habitat. While Moran and Stephenson (2000) estimated that a demersal otter trawl reduced benthos (>20cm in maximum dimensions) density by 15.5% in a single pass, Cappo et al. (1998) estimated that complete denuding of the sea bottom structure occurs after 10 - 13 trawl passes over the same area. Of equal importance are the observations of Moran and Stephenson (2000), who noted variations among trawl studies, possibly due to differences of employed ground ropes. These variations are a warning against generalizations about the impact of otter trawls on attached benthos.

As many epifaunal and infaunal organisms create structures which provide habitat for other species, summaries of these studies and their findings are included. For example, many infauna species and other bioturbators have an important role in maintaining the structure and oxygenation of muddy sediment habitats. Consequently, any

adverse effects on these organisms would presumably lead to changes in habitat complexity and community structure (Jennings and Kaiser 1998). Furthermore, the loss of biogenic epifaunal species (epibenthic habitat) increases the predation risk for juveniles of other species, thereby lowering subsequent recruitment to adult stocks (Bradstock and Gordon 1983; Walters and Juanes 1993; Jennings and Kaiser 1998). Therefore, reduction in biomass of epifaunal species may be considered a reduction or degradation of habitat in certain instances and trawling has been documented to decrease mean individual biomass of epibenthic species (Sainsbury et al. 1993; Prena et al. 1999). While it may be hard to quantify the impact this loss presents to habitat-dependent organisms, it should be noted nonetheless.

In a long-term study of Corpus Christi Bay, Texas, Flint and Younk (1983) noted that the continual minor and random disturbance, both in time and space, of channel sediments by large tanker traffic and shrimp trawling probably was sufficient to keep these communities in a state of constant disruption. This allowed the opportunists to persist more successfully than other species. The disturbed channel sites of the study, though viable, consistently had lower densities, lower numbers of species and corresponding low diversities contrasted to the lesser impacted shoal sampling sites (Flint and Younk 1983). Engel and Kvitek (1998) investigated two adjacent areas in 180m of water to determine the differences between a heavily trawled site and a lightly trawled site. They concluded that high-intensity trawling apparently reduces habitat complexity and biodiversity while simultaneously increasing opportunistic infauna and the prey of some commercial fish. The data indicated that intensive trawling significantly decreased habitat heterogeneity. All epifaunal invertebrates counted were less abundant in the highly trawled area. Bergman and Santbrink (2000) estimated direct mortality on various species of benthic megafauna from a single pass of an otter trawl (sole fishery) at between 0 - 52% for silty sediments and between 0 - 30% for sandy sediments. In general, small-sized species tend to show lower direct mortalities, when compared with larger sized species and smaller individuals of megafaunal species tend to show lower mortalities than larger-sized ones. Krost and Rumohr (1990) noted damage directly resulting from otter trawl doors. Benthic organisms were found to be reduced in number by 40 to 75% in otter board tracks, as compared to control sites. Biomass was also generally reduced. However, they found almost no differences in epibenthic species such as crustaceans. In shallow areas with densely packed sediments, inhabitants of the upper sediment layer were found to suffer most by the trawling impact.

Negligible Overall Impact?

In contrast to the above studies, there are several studies that document no significant habitat impact. Van Dolah et al. (1991) found no long-term effects of trawling on an estuarine benthic community; five months of shrimp trawling in areas previously closed to fishing were found to have no pronounced effect on the abundance, diversity, or composition of the soft bottom community when compared to nearby fished areas. They concluded that seasonal reductions in the abundance and numbers of species sampled had a much greater effect than fishing disturbance. In a power analysis of their sampling strategy, Jennings and Kaiser (1998) noted that Van Dolah et al. (1991) only considered changes in the abundance of individuals and the number of species. This assumes that the response of the infauna to trawling disturbance was unidirectional, whereas a consideration of changes in partial dominance might have been more sensitive to subtle changes in the fauna. Yet, Jennings and Kaiser (1998) stated that the results of Van Dolah et al. (1991) were plausible and that light shrimp trawls probably do not cause significant disturbance to communities in poorly sorted sediments in shallow water. Sanchez et al. (2000) determined that sporadic episodes of trawling in muddy habitats may cause relatively few changes in community composition. They found similar infaunal community changes in both fished and unfished control areas through time. Sanchez et al. (2000) also noted that the decrease in the abundance of certain species in the unfished control areas may indicate that the natural variability at the experimental site exceeds the effects of fishing disturbance. Regardless, Ball et al. (2000) commented that epifauna are generally scarce in muddy sediment habitats, and detection of fishing effects on such species has therefore been limited.

While the passage of a trawl may damage or destroy macroinfauna, Gilkinson et al. (1998) suggested that smaller infauna are resuspended or displaced by a pressure wave preceding otter trawl doors and are redeposited to the sides of the gear path. Due to a buffer effect caused by a displacement field of sediment (sand), bivalves incur

a low level of damage (5%) by the passing of a trawl door. In contrast to coarse sediment communities where the infauna are found within the top 10cm, organisms in soft mud communities can burrow up to two meters deep (Atkinson and Nash 1990). Due to their depth, it is likely that these organisms are less likely impacted by passing trawls (Jennings and Kaiser 1998), though it should be noted that the energetic costs of repeated burrow reconstruction may have long-term implications for the survivorship of individuals.

Studies documenting impacts to habitat from successive trawling are not prevalent. However, a few studies suggest that shifts in species abundance and diversity are a result of the cumulative effects of trawling. Over a longer time scale (i.e., 50 years), Ball et al. (2000) suggested that fishing disturbance may ultimately lead to an altered, but stable, community comprising a reduced number of species, and hence, diversity. Sainsbury et al. (1993; 1997) noted that composition of a multispecies fish community in Australia were at least partially habitat dependent and that historical changes in relative abundance and species composition in this region were at least in part a result of the damage inflicted on the epibenthic habitat by the demersal trawling gear.

In summary, trawling has the potential to reduce or degrade structural components and habitat complexity by removing or damaging epifauna; smoothing bedforms which reduces bottom heterogeneity; and removing structure producing organisms. Trawling may change the distribution and size of sedimentary particles; increase water column turbidity; suppress growth of primary producers; and alter nutrient cycling. The magnitude of trawling disturbance is highly variable. The ecological effect of trawling depends upon site-specific characteristics of the local ecosystem such as bottom type, water depth, community type, gear type, as well as the intensity and duration of trawling and natural disturbances. It should also be noted that there is not a direct relationship between the overall amount of trawling effort and the extent of subsequent impacts or the amount of fauna removed because trawling is aggregated and most effort occurs over seabed that has been trawled previously (Pitcher et al. 2000). Yet, several studies indicate that trawls have the potential to seriously impact sensitive habitat areas such as SAV, hardbottom, and coral reefs. In regard to hardbottom and coral reefs, it should be recognized that trawlers do not typically operate in these areas due to the potential damage their gear may incur. While trawl nets have been documented to impact coral reefs, typically resulting in lost gear (Bohnsack personal observation), these incidents are usually accidental. Partially in response to accusations of trawl activity on hardbottom habitat, a recent research effort to investigate potential impacts on the Florida Middle Ground Habitat Area of Particular Concern concluded that there was no evidence of trawl impacts or other significant fishery-related impacts to the bottom (Mallinson unpublished report). However, low-profile, patchy hardbottom or sponge habitat areas are more likely impacted from trawls due to the gear's ability to work over these habitat types without damaging the gear. Regardless, while it may be concluded that trawls have a minor overall physical impact when employed on sandy and muddy substrates, the available information does not provide sufficient detail to determine the overall or long-term effect of trawling on regional ecosystems.

RECOVERY

Recovery of substrate depends on sediment type, depth, and natural influences such as currents and bioturbation. Schoellhamer (1996) investigated sediment resuspension within Tampa Bay, a shallow estuary with fine non-cohesive material (muds absent), and found that sediment concentrations returned to pre-trawl conditions approximately 8 hours after disturbance. The cumulative effect of several trawlers operating were not investigated. DeAlteris et al. (1999) found that scars similar to those that occur from otter trawl boards disappear relatively quickly in a shallow sand environment, while those occurring in a deeper mud habitat took as long as two months to disappear. DeAlteris et al. (1999) also found that natural disturbances to mud substrate in 14m of water are rarely capable of disturbing the seabed. Therefore, recovery of fishery-related impacts in deeper water may be protracted due to the lack of natural events that help deposit sediments and fill trawl scars. Ball et al. (2000) determined that intensive demersal trawling over muddy seabeds leads to apparent long-term alteration of the seabed. Trawl tracks in muddy sediments may last up to 18 months, however, in areas of strong tidal or wave action, they are likely to disappear rapidly. Also, in areas where levels of bioturbation are high, and a regular turnover of sediment produces large numbers of mounds on the seabed, trawl tracks will be filled relatively quickly (Ball et al. 2000). Habitats in deeper water tend to recover at a slower rate. Berms and furrows generated by trawl doors generally disappeared within one year in sandy habitats in depths of approximately 120 -

146m (Schwinghamer et al. 1998; Prena et al. 1999). More dramatic is the estimate of 50 - 75 years to fill a typical trawl mark (~15cm scour depth) in deep water (>175m) by Friedlander et al. (1999). The greater the water movement, the faster the scars will be filled in (Jones 1992). Churchill (1989) and Krost et al. (1990) reported an increase in the frequency of tracks attributed to trawl doors in deeper water, presumably where water movement and natural impacts are less pronounced.

In general, few studies document recovery rates of habitat. Those that do investigate recovery usually only do so after a single treatment which does not reflect the reality of fishing impacts which are ongoing and cumulative. For example, Van Dolah et al. (1983; 1987) noted that hardbottom habitat in his trawl study recovered within one year. However, the experiment did not investigate the cumulative and repetitive effects of trawling at commercial intensities. As noted by an ICES (1995) study, due to the cumulative effects of trawling, focus on the scale of individual trawl impacts may be inadequate for estimating the importance of impacts on benthic communities. ICES (1994) stated that deep water coral banks (e.g., *Oculina varicosa*), due to their fragility, long-life spans, and infrequent recruitment, may be nearly exterminated by a single passage of a trawl and are unlikely to recover "within a foreseeable future." Likewise, SAV would also have a protracted recovery time in comparison to sediments. SAV recovery may vary by species and can be greater than two years if the rhizomes of the plant are removed (Homziak et al. 1982). Regardless, the majority of studies concur that shallow communities have proved to be resilient due to their adaptation to highly variable environmental conditions and thus, recovery is usually swift. Kaiser et al. (1996a) found epifaunal communities in 35m of water that were experimentally trawled were indistinguishable from control sites after six months. In areas of low current or great tidal exchange (e.g., deep ocean), where the benthos is not adapted to high sediment loads, the adverse effects of sediment resuspension by gear could persist for decades (Jones 1992). Recovery of small epibenthic organisms may be relatively rapid, but recovery of larger epibenthic organisms would be expected to be much slower. Though they did not discuss depth as a controlling factor, Sainsbury et al. (1993; 1997) indicated that there would be a considerable time lag after trawling ceases before recovery of large epibenthic organisms is substantial. In general, Boesch and Rosenberg (1981) predicted that recovery times for macrobenthos of temperate regions would be less than five years for shallow waters (including estuaries) and less than ten years for coastal areas of moderate depth.

MANAGEMENT RECOMMENDATIONS

The majority of management recommendations indicate that marine reserves or gear zoning may be the most effective at reducing habitat impacts. However, other specific recommendations can be extracted from several studies. Tabb (1958) recommended that otter trawls not be permitted to operate in the bait shrimp fishery due to potential impact to SAV communities. Van Dolah et al. (1987) suggested that trawls with doors attached directly to the nets would greatly reduce the bottom area damaged by trawling activities. The use of artificial reefs to protect the seabed, in particular along the perimeter of SAV habitat areas, from trawling has also been offered (Guillén et al. 1994; Ardizzone et al. 2000). The use of semi-pelagic trawls would avoid the majority of habitat impacts that demersal trawls are associated with. However, while the use of semi-pelagic nets does not significantly impact the benthos, catch efficiency may be greatly reduced. Furthermore, enforcement on the use of semi-pelagic nets remains difficult (Moran and Stephenson 2000). Carr and Milliken (1998) offered more straightforward recommendations: target certain species and modify gear appropriately; encourage the use of lighter sweeps; reduce the sea bottom available to trawlers that fish very irregular terrain; and opt for stationary gear over mobile gear.

It is suggested that where fishing effort is constrained within particular fishing grounds, and where data on fishing effort are available, studies that compare similar sites along a gradient of effort have produced the types of information on effort impact that will be required for effective habitat management (Collie et al. 1997; Auster and Langton 1999). Additionally, the use of an indicator species (e.g., quahogs) that provides a historical record of fishing disturbance events could greatly enhance the interpretation of perceived changes ascertained from samples of present-day benthic communities (Macdonald et al. 1996; Kaiser 1998). Finally, the use of tracking devices (VMS) would provide a means for identifying the most heavily fished areas and those, if any, that are presently unfished (Macdonald et al. 1996; Kaiser 1998).

Comprehensive mapping of benthic habitats may provide the necessary information to determine what areas are at risk from fishery-related impacts. Utilized in conjunction with information that details fishing effort and area, gear zoning that limits the vulnerability of sensitive habitats while minimizing economic impacts to fishery participants should be considered.

OYSTER DREDGE

An oyster dredge (Figure 5) consists of a metal rectangular frame to which a bag-shaped net of metal rings is attached. The frame's lower end is called the raking bar, and is often equipped with metal teeth used to dig up the bottom. The frame is connected to a towing cable and dragged along the seabed. Oyster dredges are widely utilized in state waters along the Gulf of Mexico, as well as the South Atlantic.

IMPACTS

Mechanical harvesting of oysters using dredges extracts both living oysters and the attached shell matrix and has been blamed for a significant proportion of the removal and degradation of oyster reef habitat (Rothschild et al. 1994; Dayton et al. 1995; Lenihan and Peterson 1998). Lenihan and Peterson (1998) observed that less than one season of oyster dredging reduced the height of restored oyster reefs by ~30%. Reduction in the height of natural oyster reefs is expected to be less than that of restored reefs because the shell matrix of natural reefs is more effectively cemented together by the progressive accumulation of settling benthic organisms, while restored reefs are initially loose piles of shell material. Regardless, it is likely that the height of natural reefs is also reduced by dredging because a large portion of extracted material from natural reefs by dredges is shell matrix. Lenihan and Peterson (1998) stated that it was probable that reduction in reef heights in a Neuse River, North Carolina estuary was due to decades of fishery-related disturbances caused by oyster dredging. At an annual removal rate of 30%, restored reefs would be completely destroyed after <4 years of harvesting. Furthermore, they determined that the height reduction of oyster reefs through fishery disturbance impacted the quality of habitat due to the seasonal bottom-water hypoxia/anoxia which caused a pattern of oyster mortality and influenced the abundance and distribution of fish and invertebrate species that utilize this temperate reef habitat (Lenihan and Peterson 1998). Their results illustrated that tall experimental reefs – those mimicking natural, ungraded reefs – were more dependable habitat for oysters and other reef organisms than short reefs – those mimicking harvest-degraded reefs – because tall reefs provided refuge above hypoxic/anoxic bottom waters. Chestnut (1955) also documented that intensive dredging over a period of years resulted in the removal of the productive layer of shell and oyster, leaving widely scattered oysters and little substrate for future crop of oysters. Glude and Landers (1953) noted that dredges mixed the sandy-mud layer and the underlying clay. Fished areas were found to be softer and have less odor of decomposition than the unfished control site. Glude and Landers (1953) also found a decrease in benthic fauna in the fished sites versus the unfished control sites.

Conversely, a study conducted by Langan (1998) which looked at the impacts oyster dredging had on benthic habitat, as well as sediment resuspension resulting from dredging activity, concluded with different results. He noted that the size frequency of oysters from the control site were biased towards older and larger specimens with poor recruitment. Oysters from the dredged site illustrated good recent recruitment, while larger specimens were not as abundant as the control site. No significant differences between the two areas were found in number, species richness, or diversity of epifaunal and infaunal invertebrates, indicating that dredge harvesting had no detectable effect on the benthic community. Sediment suspension resulting from dredging activity appeared to be localized. It should be noted that the study failed to evaluate fishing activity (number of participants, effort) on the dredged site.

RECOVERY

No information is provided in the literature in regard to recovery metrics. However, it may be noted that recovery may be protracted as fishing intensity increases.

MANAGEMENT RECOMMENDATIONS

Due to overfishing and disease, oysters may now be more economically valuable for the habitat they provide for other valued species than they are for the oyster fishery (Lenihan and Peterson 1998). Rothschild et al. (1994) suggested the establishment of broodstock sanctuaries that includes the designation of "no-fishing" restrictions in specific areas. Lenihan and Micheli (2000) also recommended the closure of some oyster reefs to harvest. Maintaining high densities of oysters on some intertidal reefs may help to preserve future oyster harvests and broodstock. Furthermore, protecting some reefs will also preserve the ecological functions that oyster reef provide such as improving water quality and providing essential recruitment, refuge, and foraging habitat for numerous marine species.

SCALLOP DREDGE (INSHORE)

Scallop dredges are similar to crab scrapes, though scallop dredges utilized in the Southeast generally do not have teeth located on the bottom bar. Scallop dredges (Figure 6) are predominately used on SAV beds where bay scallops can be efficiently harvested, and thus, are primarily limited to state waters. Popular bay scallop fisheries exist both off Florida and North Carolina. This gear, while similar, is not the same type of dredge utilized offshore to harvest calico scallops (*Argopecten gibbus*) or Atlantic sea scallops (*Placopecten magellanicus*).

IMPACTS

Though scallop dredges do not have teeth that would easily uproot SAV, studies have noted a reduction of algal and SAV biomass from their use (Fonseca et al. 1984; Bargmann et al. 1985). The reduction of SAV (*Zostera marina*) biomass was linearly related to the number of times a particular area was dredged, and the effects of dredging were proportionately greater on soft bottom than hard bottom (Fonseca et al. 1984). The Fonseca et al. (1984) study utilized an empty dredge that was 60% of the legal limit for a commercial dredge, and was not employed in conjunction with a boat as the commercial fishery does. Hand dredging was done to eliminate propeller scour which commonly occurs in shallow SAV beds. In commercial scalloping, the added dredge and scallop weight, as well as the propeller wash, could be expected to have a greater impact (Fonseca et al. 1984). In general, more damage from scallop dredging occurred to SAV in soft substrates (i.e., mud) than hard substrates (i.e., sand). In softer sediments, plants were uprooted and damage to underground plant tissues, including meristems, occurred. In harder sediments, damage was found to be generally greater for above ground parts; underground meristems were left intact and able to begin to repair shoots or produce new ones after impacts had ceased (Fonseca et al. 1984).

RECOVERY

Fonseca et al. (1984) determined that in a lightly harvested SAV area, with <25% biomass removal, recovery occurred within a year. In areas where harvesting resulted in the removal of 65% of SAV biomass, recovery was delayed for two years. After four years, preharvesting biomass levels were still not obtained. These estimates were based on termination of fishery-related impacts. Continued fishing activity would likely lead to prolonged recovery and continued degradation. Homziak et al. (1982) estimated that SAV recovery can be greater than two years if the rhizomes of the plant are removed.

MANAGEMENT RECOMMENDATIONS

Due to the importance of SAV beds as a nursery area to other species, loss of eelgrass meadows should be avoided. Fonseca et al. (1984) suggested that harvest area rotation may minimize habitat impact.

SCALLOP DREDGE (OFFSHORE)

Scallop dredges (Figure 7) utilized to harvest calico or sea scallops consist of a metal frame that supports tickler chains and a metal ring bag that collects the shellfish. Though not widely utilized in the Southeast, the gear has been included in this review due to their inclusion as an approved gear in the South Atlantic. The majority of studies on scallop dredge impacts originate from areas with extensive scallop fisheries such as the northwest and northeast Atlantic.

IMPACTS

Due to the potential for the gear to have considerable weight and the fact that it is dragged along the bottom, habitat impacts are expected to occur. Drew and Larsen (1994) estimated that a scallop dredge maximum cutting depth would be 40 - 150mm. Kaiser et al. (1996a) found that scallop dredging greatly reduced the abundance of most species, causing significant changes in the community. It was noted that a large proportion of some animals (such as echinoderms) were not captured or passed through the mesh of the gear. The scallop dredge catches contained a low proportion of non-target species which indicates that the belly rings allow the bycatch to escape. However, the study did not investigate the extent of damage/injury to organisms that were not captured. Likewise, Collie et al. (1997) found areas on Georges Bank that were impacted by scallop dredges to have lower species diversity, lower biomass of fauna, and dominated by hard-shelled bivalves, echinoderms, and scavenging decapods. Areas less impacted by dredges had higher diversity indices. However, it should be noted that portions of Georges Bank consist of cobble habitat which is encrusted with a diverse array of epibenthic species. Perhaps more applicable to the areas in the Southeast where calico scallops are harvested off North Carolina and Florida, would be a study conducted by Butcher et al. (1981), who determined that scallop dredges had little or no environmental effect when they were used on large-grained, firm sand bottom that was shaped in roughly parallel ridges. The area in this study was also noted to be a fairly uniform, low species diversity community. Turbidity caused by the turbulence of the dredge quickly dissipated due to the nature of the substrate. Additionally, Jolley (1972) found no detrimental dredging effects on sand substrates. Yet, there is a potential for dredges to impact coral adjacent to scallop beds, especially the scallop grounds which occur in close proximity to the Oculina Bank off eastern Florida. Should a scallop dredge impact *Oculina* coral, there would be severe results, similar to the conclusions reached by ICES (1994) for trawls. This study determined that deep water coral banks such as those composed of *Oculina varicosa*, due to their fragility, long-life spans, slow growth, and infrequent recruitment, may be nearly exterminated by a single passage of a trawl. Recovery of this habitat area, "within a foreseeable future," is unlikely (ICES 1994).

RECOVERY

Collie et al. (1997) found that biogenic epifauna on Georges Bank showed signs of recovery after two years at a site that was dredged for scallops and then closed to fishing. The areas in the Southeast that are worked by scallop dredges largely consist of sandy substrates, therefore recovery may occur in a shorter timeframe.

MANAGEMENT RECOMMENDATIONS

No specific or applicable management recommendations are offered in the literature. As this is not a prominent gear type, no broad management measures may be necessary. Rather, specific management measures, such as the recent expansion of the Oculina Bank Habitat Area of Particular Concern where fishing by bottom-tending gear is prohibited, should be offered when at-risk habitat areas are identified.

SKIMMER TRAWL

Skimmer trawls are positioned along the side of a boat and pushed through the water to harvest shrimp. Two nets are typically used, one on each side of the boat. Skimmer trawls (Figure 8) are supported by a tubular metal frame that skims over the bottom on a weighted metal shoe or skid. Tickler chains are also utilized along the base of the net. Because of the construction attributes of this gear type, skimmer trawls are generally restricted to water 3.05m (10ft) or less which would limit them to state waters.

IMPACTS

Skimmer trawls work on mud bottoms in water generally 3.05m (10ft) or less. The weighted shoe and tickler chains impact the bottom, resulting in sediment resuspension. Skimmer trawls may cause bottom damage due to improperly tuned or poorly designed gear (skids and bullets) or prop damage in shallow areas (Steele 1994). Furthermore, because skimmer trawls are used in shallow water, they may have a detrimental impact on critical nursery areas such as the marsh/water interface, SAV, or other sensitive submerged habitats. However, skimmer trawls are expected to impact the bottom less than otter trawls due to the absence of doors (Nelson 1993; Steele 1993).

Coale et al. (1994) believed that the skimmer trawl would not have any greater effects on SAV than the otter trawl. They found it doubtful that the inside weight and outer shoe of the skimmer trawl would cause greater detrimental effects to the benthos than the heavy doors of an otter trawl. Based on underwater observations, Coale et al. (1994) suggested that the weight and shoe combination may be less-damaging than otter trawls. However, habitat such as sponges and SAV are cut off by tickler chains and lead lines whereas otter trawl doors can dig in and tear up the bottom. Given the difference in the amount of area covered by each on normal tows, Kennedy, Jr. (1993) found it doubtful that there would be much difference in the amount of habitat loss between skimmer trawls and otter trawls.

RECOVERY

No information relative to habitat recovery from skimmer trawl impacts is provided in the literature.

MANAGEMENT RECOMMENDATIONS

Kennedy, Jr. (1993) recommended that the use of skimmer trawls in Florida should be restricted to those areas currently approved for otter trawls. Due to the associated impacts to SAV, a prudent recommendation would be to limit skimmer trawl use to non-vegetated substrates.

STATIC GEAR

It has been noted by Rogers et al. (1998) that there are few accounts of the physical contact of static gear having measurable effects on benthic biota, as the area of seabed affected by each gear is almost insignificant compared to the widespread effects of mobile gear. Nevertheless, static gear can impact habitat and needs to be evaluated.

CHANNEL NET

Channel nets are fixed to pilings, docks, or shore installation and utilize current flow to capture shrimp, therefore, channel nets are limited to use within state waters.

IMPACTS

Though impacts of channel nets were not discussed specifically, it may be inferred from Higman (1952) that channel nets have negligible impact on habitat due to catch composition and the lack of interaction with the benthos.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

GILLNET & TRAMMEL NET

Gillnets (Figure 9) consist of a wall of netting set in a straight line, equipped with weights at the bottom and floats at the top, and is usually anchored at each end. As fish swim through the virtually invisible monofilament netting, they become entangled when their gills are caught in the mesh, hence the name. Gillnets may be fixed to the bottom (sink net) or set midwater or near the surface to fish for pelagic species. A trammel net (Figure 10) is made up of two or more panels suspended from a float line and attached to a single lead line. The outer panel(s) are of a larger mesh size than the inner panel. Fish swim through the outer panel and hit the inner panel which carries it through the other outer panel, creating a bag and trapping the fish. Smaller and larger fish become wedged, gilled, or tangled. Gillnets are widely used in numerous fisheries, both in state waters and in Federal waters. Trammel nets are primarily used in state waters, though they are an authorized gear in the Caribbean for both the spiny lobster and shallow water reef fish fisheries.

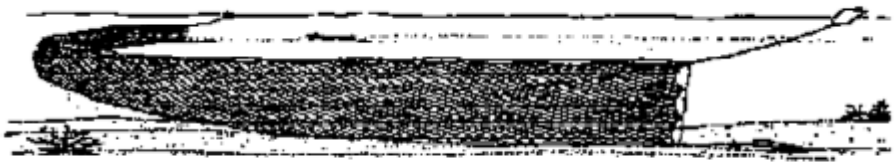


FIGURE 9. GILLNET (West et al. 1994)

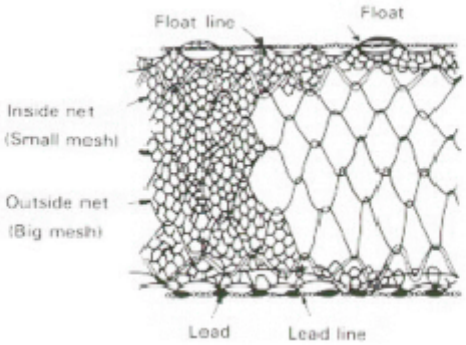


FIGURE 10. TRAMMEL NET (Yusung Industrial Co., Ltd.)

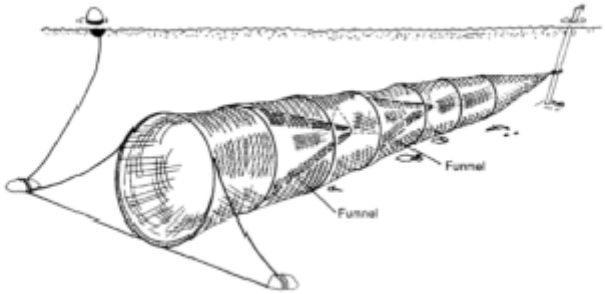


FIGURE 11. HOOP NET (Nielsen and Johnson 1983)

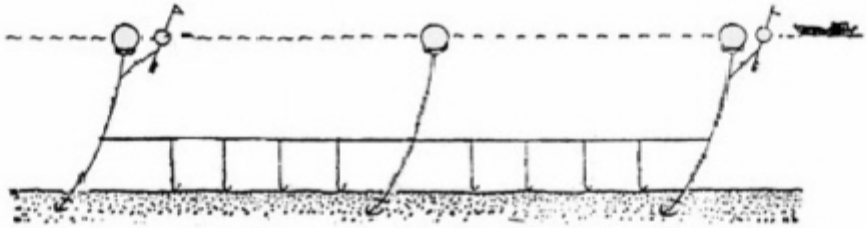


FIGURE 12. BOTTOM LONGLINE

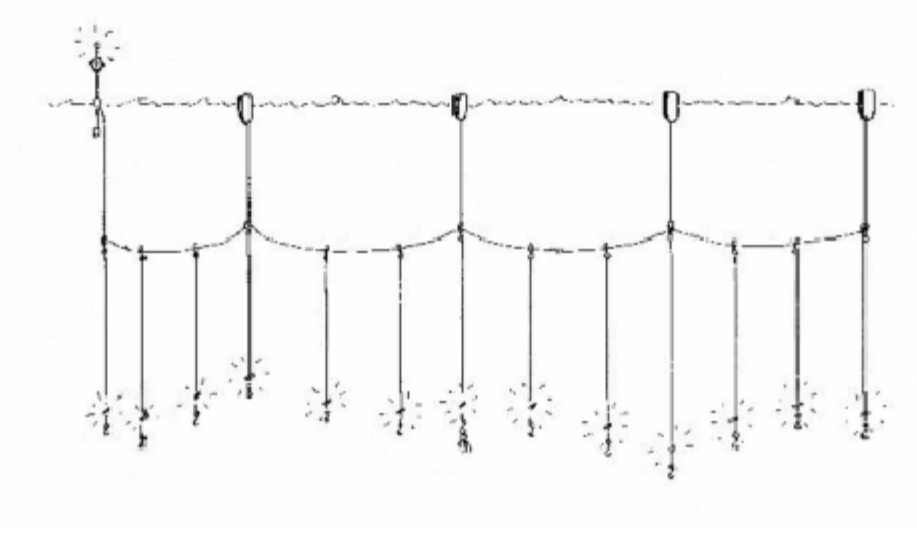


FIGURE 13. PELAGIC LONGLINE (Stephen Willoughby)

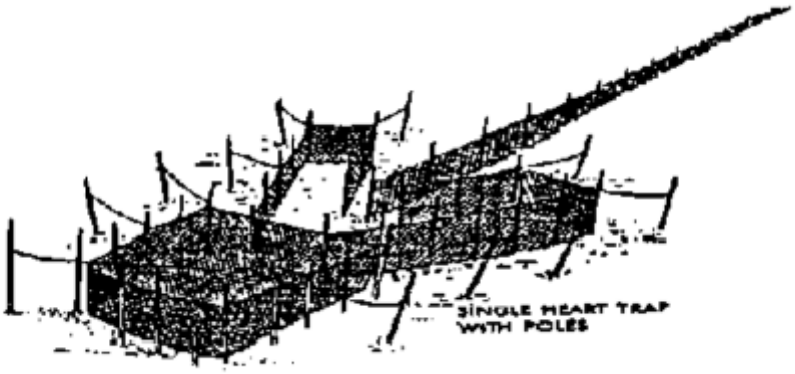


FIGURE 14. POUND NET (West et al. 1994)

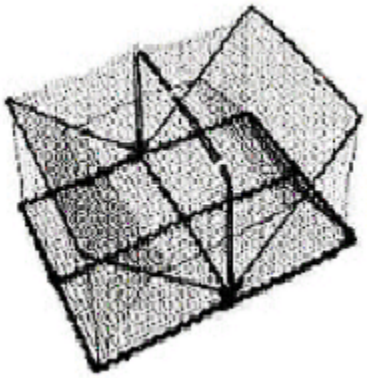


FIGURE 15. FISH TRAP

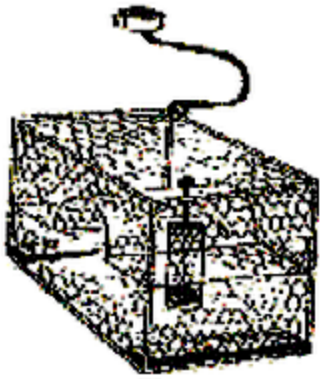


FIGURE 16. CRAB POT

FIGURE 17. COLLAPSIBLE CRAB TRAP

IMPACTS

The majority of the studies that have investigated impacts of fixed gillnets have determined that they have a minimal effect on the benthos (Carr 1988; ICES 1991; ICES 1995; Kaiser et al. 1996b). An ASMFC (2000) report determined that impacts to SAV from gillnets would be minimal. Likewise, West et al. (1994) stated that there was no evidence that sink net (gillnet) activities contributed importantly to bottom habitat disturbance. However, Carr (1988) noted that ghost gillnets in the Gulf of Maine could become entangled in rough bottom. He observed one net that had its leadline and floatline twisted around each other and tightly stretched between boulders. Furthermore, Williamson (1998) noted that gillnets can snag and break benthic structures. Gomez et al. (1987) noted that gill nets set near reefs occasionally results in accidental snarring often resulting in damage to coral. Bottom set gillnets have led to habitat destruction in different regions (Jennings and Polunin 1996). Bottom gillnets set over coral may cause negative impacts as the weighted lines at the base of the net often become entangled with branching and foliaceous corals. As the nets are retrieved, the corals are broken (Öhman et al. 1993). This observation has also been noted in a study by Munro et al. (1987), which documented that reefs are frequently damaged by the hauling of set (gill) nets, and the problem has been exacerbated by the use of mechanical net haulers or power blocks.

Aside from the potential impacts cited on coral reef communities, the available studies indicate that habitat degradation from gillnets is minor. Several studies note that lost gillnets are quickly incorporated by marine species. Cooper et al. (1988) found ghost gillnets in the Gulf of Maine covered with a heavy filamentous growth, exceeding 75% coverage on some nets. Anemones, stalked ascidians and sponges were attached to and growing to the net float lines (Carr et al. 1985; Cooper et al. 1988). Erzini et al. (1997) found that lost trammel nets and gill nets in shallow water (15 - 18m) on rocky habitat (analogous to coral reefs and hardbottom habitat) were colonized by various species, primarily macrophytes, which after three months completely blocked the meshes of some parts of the nets. Some netting would contact reef habitat, becoming heavily overgrown and eventually blended into the background. After a year, most of the netting was destroyed; those remnants that remained were completely colonized by biota (Erzini et al. 1997). Erzini et al. (1997) also noted that the nets eventually became incorporated into the reefs, acting as a base for many colonizing plants and animals. The colonized nets then provided a complex habitat which was attractive to many organisms. For example, large schools of juvenile fish were often observed in the vicinity of these heavily colonized nets, which may provide a safe haven from predators. Johnson (1990) and Gerrodette et al. (1987) noted that as gillnets tend to collapse and "roll up" relatively quickly, they may form a better substrate for marine growths and thereby attract fish and other predators which may get entangled, ultimately causing the net to sink. Therefore, one may assume that gillnets may be more of a ghostfishing problem and entanglement hazard to marine life than as an impact to habitat.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from these gear types, no conclusions on recovery or management recommendations are offered.

HOOP NET

A hoop net (Figure 11) is a cone-shaped or flat net which may or may not have throats and flues stretched over a series of rings or hoops for support. The net is set by securing the cod or tapered end to a post or anchored to the bottom. The net is played out with the current until fully extended, and then is allowed to settle to the bottom. The net is marked with a buoy for easy retrieval and identification purposes. The duration of time that a hoop net is set depends on the same factors that influence the duration of the set of a gill net and should be determined in a similar fashion. To harvest, the hoop net is raised at the cod end and the fish are removed.

IMPACTS

While there are no studies that document the effect of hoop nets on habitat, due to its use primarily on flat bottoms the gear probably has less of an impact than traps.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

LONGLINE

Longlines use baited hooks on offshoots (gangions or leaders) of a single main line to catch fish at various levels depending on the targeted species. The line can be anchored at the bottom (Figure 12) in areas too rough for trawling or to target reef associated species, or set adrift, suspended by floats (Figure 13) to target swordfish and sharks. Longlines are widely utilized in numerous fisheries throughout the Southeast Region.

IMPACTS

When a vessel is retrieving a bottom longline it may be dragged across the bottom for some distance. The substrate penetration, if there is any, would not be expected to exceed the breadth of the fishhook, which is rarely more than 50mm (Drew and Larsen 1994). More importantly is the potential effect of the bottom longline itself, especially when the gear is employed in the vicinity of complex vertical habitat such as sponges, gorgonians, and corals. Observations of halibut longline gear off Alaska included in a North Pacific Fishery Management Council Environmental Impact Statement (NPFMC 1992) provide some insight into the potential interactions longline gear may have with the benthos. During the retrieval process of longline gear, the line was noted to sweep the bottom for considerable distances before lifting off the bottom. It snagged on whatever objects were in its path, including rocks and corals. Smaller rocks were upended and hard corals were broken, though soft corals appeared unaffected by the passing line. Invertebrates and other light weight objects were dislodged and passed over or under the line. Fish were observed to move the groundline numerous feet along the bottom and up into the water column during escape runs, disturbing objects in their path. This line motion has been noted for distances of 15.2m (50ft) or more on either side of the hooked fish. Based on these observations, it is logical to assume that longline gear would have a minor impact to sandy or muddy habitat areas. However, due to the vertical relief that hardbottom and coral reef habitats provide, it would be expected that longline gear may become entangled, resulting in potential impacts to habitat. Due to a lack of interaction with the benthos, pelagic longlines would have a negligible habitat impact.

RECOVERY

Due to the lack of sufficient scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery are offered.

MANAGEMENT RECOMMENDATIONS
Due to the potential entanglement impacts associated with bottom longlines, excluding their use in the vicinity of sensitive benthic habitat such as coral reefs would be an appropriate management measure.

POUND NET

A pound net (Figure 14) consists of a fence constructed of netting that runs perpendicular to shore which directs fish to swim voluntarily into successive enclosures known as the heart, pound, or pocket. Pound nets are exclusively utilized in state waters.

IMPACTS
An ASMFC (2000) report determined that impacts to SAV from pound nets are expected to be minimal, unless the net is constructed directly on SAV. West et al. (1994) also stated that pound nets do not contribute to benthic disturbance. Due to the limited amount of space a pound net may impact, it is expected that pound nets have minimal impact on habitat.

RECOVERY & MANAGEMENT RECOMMENDATIONS
Due to the lack of sufficient scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

TRAP & POT

Traps and pots (Figures 15 - 17) are rigid devices, often designed specifically for one species, used to entrap finfish or invertebrates. Generally baited and equipped with one or more funnel openings, they are left unattended for some time before retrieval. Traps and pots are weighted to rest on the bottom, marked with buoys at the surface, and are sometimes attached to numerous other traps to one long line called a trot line. Traps and pots are widely used on a variety of habitats in both state and Federal waters to harvest species such as lobster, blue crabs, golden crabs, stone crabs, black sea bass, snapper, and grouper. Wire-mesh fish traps are one of the principal fishing gears used in coral reef areas in the Caribbean (Appledorn 2000).

IMPACTS
Due to their use to harvest species associated with coral and hardbottom habitat, traps and pots have been identified to impact and degrade habitat. Gomez et al. (1987) noted the incidental breakage of corals on which traps may fall or settle constitute the destructive effects of this gear. Within the Virgin Islands State Park, Garrison (1998) found 86% of the fish traps were set on organisms (live coral, soft coral, SAV) living on the sea floor. Damage to the live substrate has far-reaching negative effects on the marine ecosystem because the available amount of shelter and food often decreases as damage increases. Another study conducted by Garrison (1997) had similar results, as 82% of traps rested directly on live substrate, with 17% resting on stony corals. Hunt and Matthews (1999) found that lobster and stone crab traps reduce the abundance of gorgonian colonies from rope entanglement. Furthermore, seagrass smothering occurs from trap placement on SAV beds, resulting in SAV "halos." Van der Knapp (1993) noted that fish traps set on staghorn coral easily damaged the coral. It appeared that in all observed cases of injury due to traps, the staghorn coral regenerated completely, although the time for regeneration varied from branch to branch. The greatest impact noted from the setting of traps was observed when the point of the trap's frame ran into coral formations. Several different species of coral were observed to suffer damage from fish traps. Observations of at least one damaged coral specimen noted that algae growth prevented regeneration in the damaged portion of the coral. Additionally, complete deterioration of a vase sponge was observed after it had been severely damaged by a trap. Traps are not placed randomly, rather they are fished in specific areas multiple times before fishing activity moves to other grounds. Therefore, trap damage will be concentrated (cumulative effect) in particular areas rather than be uniform over all coral reef habitat.

In a recent study, Appledorn et al. (2000) commented that traps may physically damage live organisms, such as corals, gorgonians, and sponges, which provide structure and in some cases, nutrition for reef fish and invertebrates. Damage may include flattening of habitats, particularly by breaking branching corals and gorgonians; injury may lead to reduced growth rates or death, either directly or through subsequent algal overgrowth or disease infection. During initial hauling, a trap may be dragged over more substrate until it lifts off the bottom. Traps set in trotlines can cause further damage from the trotline being dragged across the bottom, potentially shearing off at their base those organisms most important in providing topographic complexity. Traps that are lost or set unbuoyed are often recovered by dragging a grappling hook across the bottom. This practice can result in dragging induced damage from all components (grappling hook, trap, trotline). The area swept by trotlines upon recovery is orders of magnitude greater than the cumulative area of the traps themselves. Appledorn et al. (2000) documented that single-buoyed fish traps off La Parguera, Puerto Rico, have an impact footprint of approximately $1m^2$ on hardbottom or reef. Of the traps investigated in the study, 44% were set on hardbottom or reef, resulting in 23% damage to coral colonies ($70cm^2$ average), 34% damage to gorgonian colonies ($56cm^2$ average), and 30% damage to sponges, though sponges were less frequently impacted due to their patchy distribution. Trap hauling resulted in 30% of the traps inflicting additional damage to the substrate.

In a similar study focusing on fish trap impacts conducted off St. Thomas, U.S.V.I., by Quandt (1999), 40% of all traps investigated were found to be resting on reef substrate. On average, 4.98% of all hard corals and 47.17% of all gorgonians were damaged; tissue damage averaged 20.03% to each gorgonian. Secondary impacts, such as trap hauling and movement due to natural disturbances were not investigated. However, the effects of pulling a string of two or more traps would most likely be much greater than one trap alone.

Eno et al. (1996) found pots that landed on, or were hauled through beds of bryozoans caused physical damage to the brittle colonies. It was noted that several species of sea pens bent in response to the pressure wave created by a descending pot and lay flat on the seabed. When uprooted, the sea pens were able to reestablish themselves in the sediment. A species of sea fan also was found to be flexible and specimens were not severely damaged when pots were hauled over them. This suggests that in some instances the direct contact of certain gears may not be the primary cause of mortality, rather the frequency and intensity may be more important. Additionally, Sutherland et al. (1983) cited little apparent damage to reef habitats inflicted from fish traps off Florida. The study found four derelict traps sitting atop high profile reefs with four other traps observed within a live-bottom area. There was no visual evidence that traps on the high profile reef killed or injured corals or sponges. One uprooted gorgonian was observed atop a ghost trap in a live-bottom area. However, these observations were made on randomly located derelict traps. Thus, the primary impacts that may occur during deployment and recovery could not be evaluated.

RECOVERY

Recovery is dependent on the type of habitat the trap is deployed on and the amount of inflicted damage. A study (Mascarelli and Bunkley-Williams *in press*) evaluated that only 30% of corals recovered from damage after 120 days, while some damage was expected to be permanent. It would also be expected that impacted corals have varying recovery time depending on individual species. Van der Knapp (1993) observed full gorgonian recovery from trap impacts within a month.

MANAGEMENT RECOMMENDATIONS

While it appears prudent to not deploy traps on coral habitat, that recommendation may be difficult to enforce. To limit trap impacts, Stewart (1999) advised that traps should not be weighted any more than is needed for them to land upright on the sea bed. Limiting the number of traps in a trotline would limit the amount of documented habitat degradation that occurs from recovery operations.

OTHER GEAR

ALLOWABLE CHEMICAL

Collectors of live tropical reef fish commonly employ anesthetics such as quinaldine. Quinaldine (2-methyl-quinoline, $C_{10}H_9N$) is the cheapest and most available of several substituted quinolines (Goldstein 1973).

IMPACTS

As a result of using this compound near corals where tropical species shelter, there may be residual effects which was discussed in a study by Japp and Wheaton (1975). Short-term impacts of quinaldine include increased flocculent mucus production, retraction of polyps and failure to reexpand with a five minute observation period, and tissue discoloration in certain species. At both study sites, octocorals were found to suffer no long-term impacts. However, a minority of Scleractinians displayed minor damage, including mild discoloration and small patches of dead tissue, three months after quinaldine treatment. Two of these specimens degraded to poor condition or displayed areas of dead tissue more than six months after initial treatment. Overall, Japp and Wheaton (1975) determined that quinaldine exposure resulted in minimal damage to corals.

RECOVERY

As noted in Japp and Wheaton (1975), impacts appear to be temporary.

MANAGEMENT RECOMMENDATIONS

Due to the short-term impacts this fishing method introduces, as well as the limited nature of the fishery itself, no management recommendations are offered.

BARRIER NET

Barrier nets are used in conjunction with small tropical nets or slurpguns to collect tropical aquarium species. The net is deployed to surround a coral head or outcropping and may or may not have a pocket or bag that fish are "herded" into for capture. Barrier nets may be utilized by tropical fish collectors in both state and Federal waters.

IMPACTS

The American MarineLife Dealers Association conducted a survey (Tullock and Resor 1996) that focused on tropical collection practices. The survey defined a sustainable fishing practice as one that a) does not cause physical damage to the reef environment; b) does not impair the captured specimen's longevity in a properly maintained aquarium environment; and c) does not damage non-target species such as coral polyps, other invertebrates, or non-aquarium fish. The survey concluded that barrier nets were a sustainable fishing practice. However, a study conducted by Öhman et al. (1993) summarized that moxy nets, a type of barrier net that is used in other regions to collect ornamental fish species, may break corals during their use. However, it is likely that damage inflicted by barrier nets would be infrequent and incidental in nature, and therefore, the gear would have a negligible effect on habitat.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

CASTNET

Used to capture baitfish and shrimp, castnets (Figure 18) are circular nets with a weighted skirt that is thrown over a schooling target. Castnets are primarily used in shallow areas such as estuaries, though they may be used to catch baitfish offshore in Federal waters.

FIGURE 18. CAST NET (University of Washington, APL)

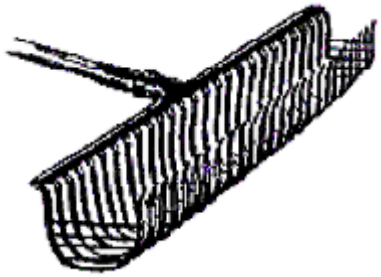
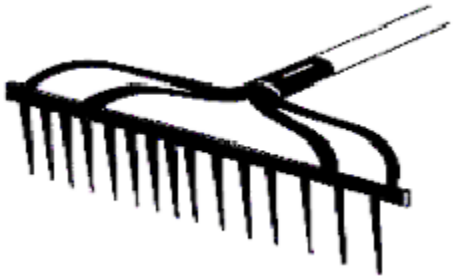
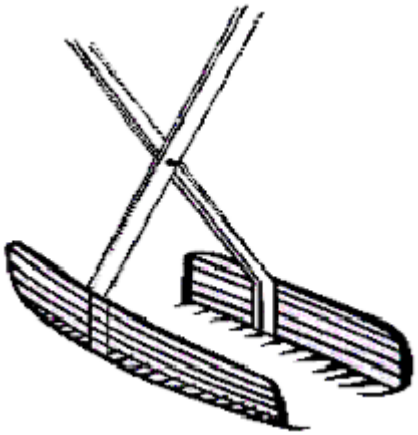


FIGURE 19. BULL RAKE FIGURE



20. HAND RAKE FIGURE



21. OYSTER TONGS

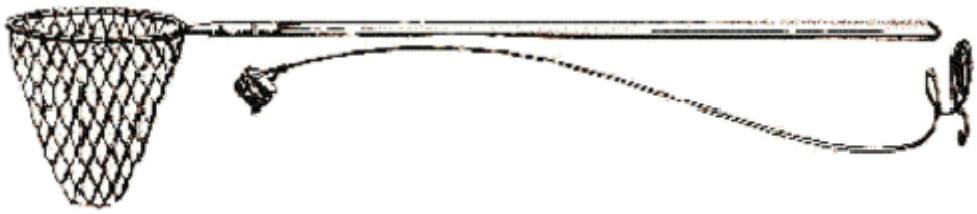


FIGURE 22. DIPNET

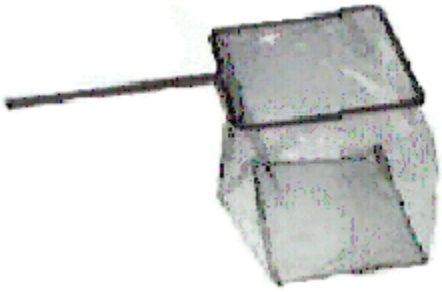


FIGURE 23. TROPICAL FISH NET

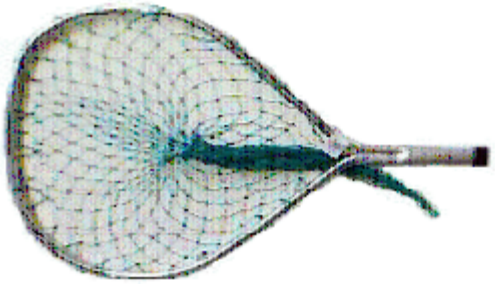


FIGURE 24. LOBSTER/LANDING NET

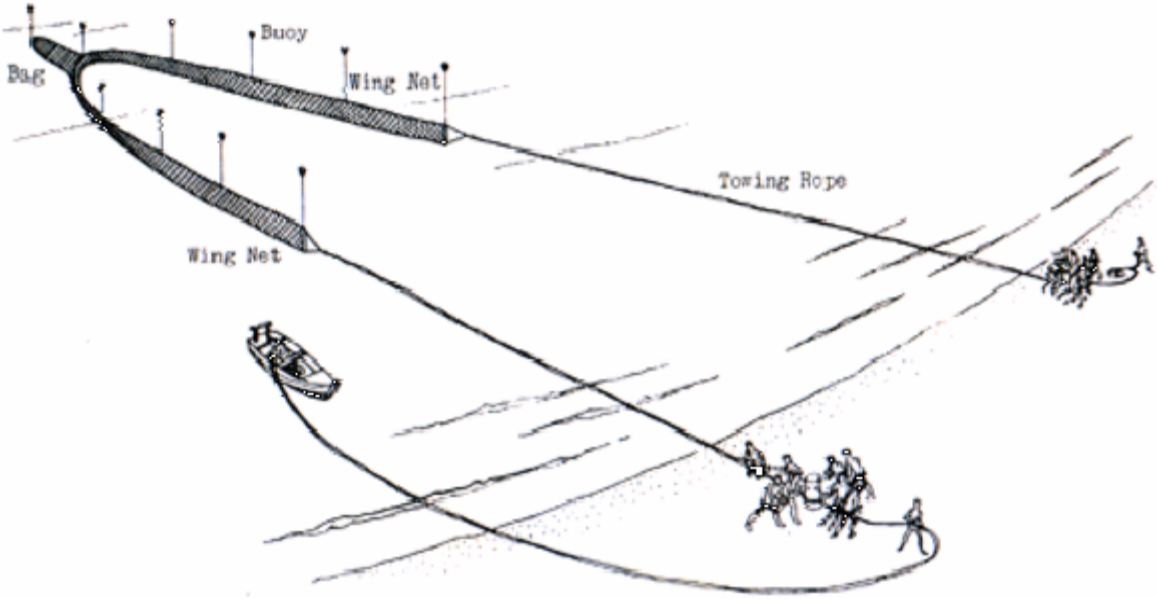


FIGURE 25. BEACH HAUL SEINE (Amita Company)

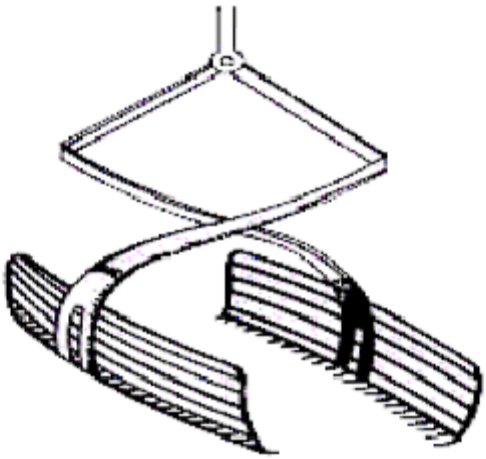


FIGURE 26. PATENT TONGS

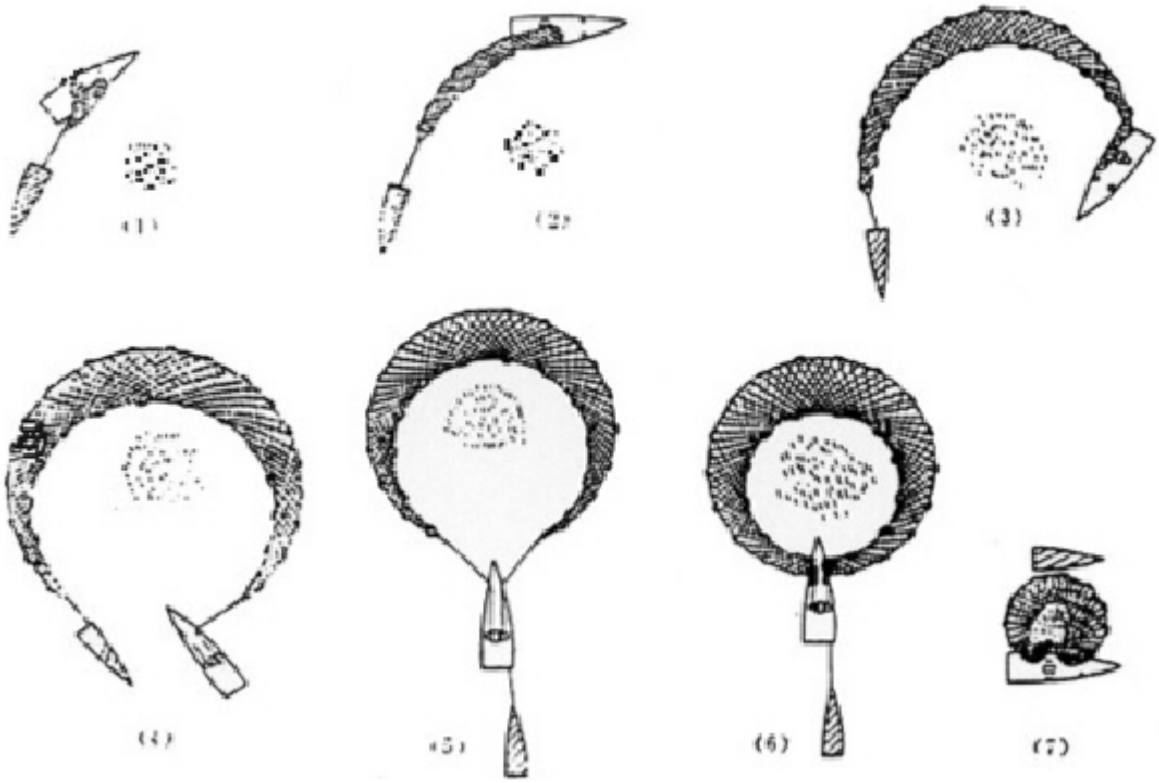


FIGURE 27. PURSE SEINE (University of Washington, APL)

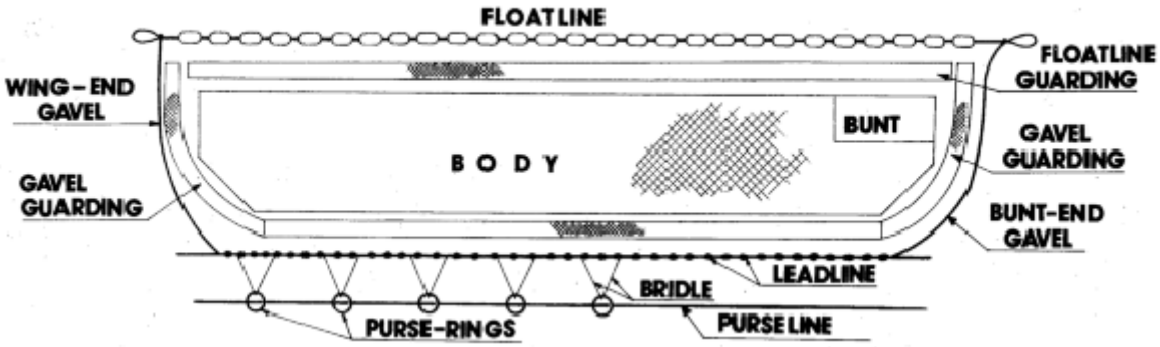


FIGURE 28. COMPONENTS OF A PURSE SEINE NET (Ben-Yami 1987)

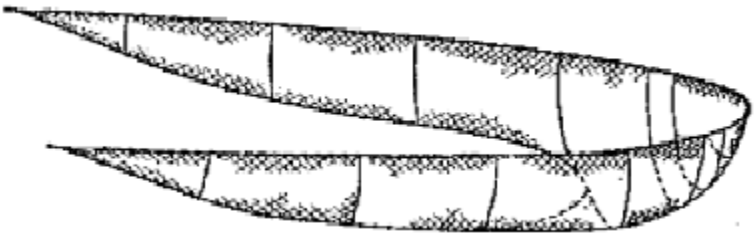


FIGURE 29. LAMPARA NET

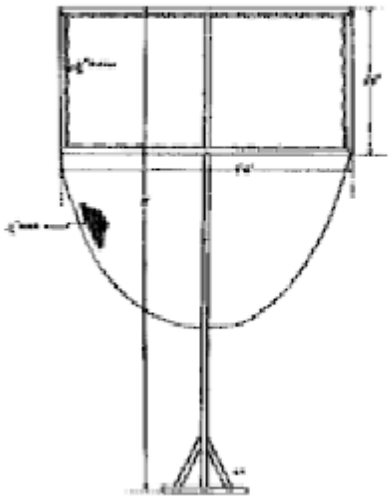


FIGURE 30. PUSHNET (De Sylva 1954)

FIGURE 31. SLURP GUN

FIGURE 32. SNARE

FIGURE 33. SPEARGUN (Riffe International)

IMPACTS

Castnets have the potential to dislodge organisms or become entangled if utilized over heavily encrusted substrates. Observations by the author have noted numerous castnets entangled amongst sponges and other growth around rough bottom. However, a study conducted by DeSylva (1954) determined that castnets have no detrimental effect on habitat.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of documented habitat impacts, no conclusions on recovery or management recommendations are offered.

CLAM KICKING

Clam kicking is a mechanical form of clam harvest primarily practiced in the state waters of North Carolina. The practice involves the modification of boat engines in such a way as to direct the propeller wash downwards instead of backwards. The propeller wash is sufficiently powerful in shallow water to suspend bottom sediments and clams into a plume in the water column, which allows clams to be collected in a trawl net towed behind the boat (Peterson et al. 1987a).

IMPACTS

Several studies have noted that the practice of clam kicking reduces algal and SAV biomass (Fonseca et al. 1984; Bargmann et al. 1985; Peterson et al. 1987a). Reduction of SAV biomass was noted to increase with harvest intensity. Intense clam kicking treatments reduced SAV biomass by approximately 65% (Peterson et al. 1987a). Because of the importance of SAV to coastal fisheries and estuarine productivity, Peterson et al. (1987a) noted that intense clam kicking could have long-lasting and serious impacts on many commercially important fisheries.

However, clam harvesting had no detectable effect on the abundance of small benthic invertebrates and outside of SAV habitat, clam kicking does not appear to have any serious negative impacts on parameters of ecological value (Peterson et al. 1987a).

RECOVERY

SAV recovery can be greater than two years if the rhizomes of the plant are removed (Homziak et al. 1982; Peterson et al. 1987a). Peterson et al. (1987a) observed that SAV had yet to recover after four years of an intense clam kicking treatment. Although Peterson et al. (1987a) designated their heavier clam kicking treatment as “intense,” they conceded that it probably falls well short of the effort that commercial clammers would apply to a productive SAV bed.

MANAGEMENT RECOMMENDATIONS

Limit the intensity of clam fishing in SAV habitat would probably be beneficial. Peterson et al. (1987a) offered that a restriction of mechanical clam harvesters to unvegetated bottoms may be a suitable mechanism to minimize habitat damage.

CLAM RAKE, SCALLOP RAKE, SPONGE RAKE, & OYSTER TONG

Rakes (Figures 19, 20) are used to harvest shellfish and sponges from shallow areas such as bays and estuaries. Oyster tongs (Figure 21), similar to two rakes fastened together and facing each other like scissors, are used by fishermen from the deck of a boat. As these gears are limited by water depth, they are exclusively utilized in state waters.

IMPACTS

Lenihan and Micheli (2000) reported that the harvest of shellfish utilizing clam rakes and oyster tongs significantly reduce oyster populations on intertidal oyster reefs. Both types of shellfish harvesting, applied separately or together, reduced the densities of live oysters by 50-80% compared with the densities of unharvested oyster reefs. While oysters are removed, Rothschild et al. (1994) concluded that hand tongs probably have a minor effect on the actual oyster bar structure.

Peterson et al. (1987b) compared the impacts of two types of clam rakes on SAV biomass. The bull rake removed over 89% of shoots and 83% of roots and rhizomes in a completely raked area while the pea digger removed 55% of shoots and 37% of roots and rhizomes. Loss or impact on SAV by bull rake was estimated to be double the impact of the smaller pea digger rake. Peterson et al. (1987a) found raking with a pea digger rake reduced SAV biomass by approximately 25%. An earlier study conducted by Glude and Landers (1953) noted that bull rakes and clam tongs mixed the sandy-mud layer and the underlying clay. Fished areas were also softer and had less odor of decomposition than the unfished control site. A decrease in benthic fauna was noted in the fished sites versus the unfished control sites.

Sponges are an important fishery in the Florida Keys and along the west coast of Florida (NOAA 1996). Sponges are dominant organisms in deepwater passes and along hardbottom habitat communities. Sponges create vertical habitat which provides shelter and forage opportunities for other invertebrates and tropical fish species. The fishery in the Keys typically employs a four-pronged iron rake attached to the end of a 5 - 7m pole which hooks the sponges from the bottom. While no studies document the extent of habitat damage from this gear type, it may be concluded that the harvest of sponges directly reduces the amount of available habitat, and thus may present a negative localized impact.

RECOVERY

Peterson et al. (1987a) found that SAV biomass recovered to equal and even exceeded expected values within one year.

MANAGEMENT RECOMMENDATIONS

Lenihan and Micheli (2000) recommended the closure of some oyster reefs to shellfish harvest. Maintaining high densities of oysters on some intertidal reefs may help to preserve future oyster harvests and broodstock. Furthermore, protecting some reefs will also preserve the ecological functions that oyster reef provide such as improving water quality and providing essential recruitment, refuge, and foraging habitat for numerous marine species. Due to the extensive habitat that sponges provide, further ecological study on the directed harvest of these organisms should be conducted.

DIPNET & BULLY NET

Widely utilized to catch baitfish, crabs, or lobster, varieties of dipnets (Figure 22) consist of a long pole with a bag of netting of varying mesh size that are lowered into the water. Dipnets may also be employed to capture tropical

reef fish (Figure 23), though these utilize a short handle and very fine mesh. Additionally, landing nets or hand bully nets (Figure 24) used to capture lobster can be considered a form of dipnet. Varieties of dipnets may be used both in state and Federal waters.

IMPACTS

DeSylva (1954) determined that dipnets have no detrimental effect on habitat. However, the use of small dipnets (i.e., tropical fish nets and lobster hand bully nets) may result in minor isolated impacts to coral species as individuals attempt to capture specimens (Barnette personal observation).

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

HAND HARVEST

Hand harvest describes activities that capture numerous species such as lobster, scallops, stone crabs, conch, and other invertebrates by hand.

IMPACTS

As many small biogenic structures occur on the sediment surface, even gentle handling by divers can destroy them easily. Movement by divers were observed to cause demersal zooplankters to exhibit escape responses (Auster and Langton 1999). A study that assessed recreational SCUBA activity in the US Caribbean (Garcia-Moliner et al. 2000) concluded that approximately 2% of the total recreational divers in the USVI and 1.9% of the total recreational divers in Puerto Rico were lobstering. Potential impact of approximately 13,532 units occurred in the USVI and 14,946 units occurred in Puerto Rico. In this study, impact units consisted of two hands and two feet (4 units per diver) and impact was broadly defined as ranging from touching coral with hands to the resuspension of sediment by fins. No assessment of habitat degradation or long-term impacts was discussed. Divers pursuing lobster along coral or hardbottom communities have been observed to impact gorgonians and other encrusting organisms (Barnette unpublished observations).

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

HARPOON

Harpoons, thrown from the decks of a vessel, are utilized to target swordfish and tuna.

IMPACTS

As this gear is employed to harvest pelagic species, there is no contact with the benthos and, thus, no impact to habitat.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the nature of this fishery and lack of physical habitat impacts, no conclusions on recovery or management recommendations are offered.

HAUL SEINE & BEACH SEINE

A haul seine (Figure 25) is an active fishing system that traps fish by encircling them with a long fence-like wall of webbing. It is made of strong netting hung from a float line on the surface and held near the bottom by a lead line. They are fished either along the shoreline (beach seine) where they are deployed in a semi-circle to trap fish between shore and net or, more typically, fish are encircled away from shore, worked into an even smaller pocket of net and lifted onto a boat for culling (Sadzinski et al. 1996). The use of this gear is limited to state waters.

IMPACTS

Sadzinski et al. (1996) found no detectable effects from haul seining on SAV. However, possible damage from haul seining to sexual reproduction, such as flower shearing, was not examined. There are possible long-term or cumulative impacts at established haul-out sites, resulting in loss of SAV biomass (Orth personal communication). As the seine is generally used in flat benthic areas to prevent the net becoming damaged, in most cases the impact from seines would be expected to be minor and temporary.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

HOOK AND LINE, HANDLINE, BANDIT GEAR, BUOY GEAR, & ROD AND REEL

These gear types are widely utilized by commercial and recreational fishermen over a variety of estuarine, nearshore, and marine habitats. Hook and line may be employed over reef habitat or trolled in pursuit of pelagic species in both state and Federal waters.

IMPACTS

Few studies have focused on physical habitat impacts from these gear types. Impacts may include entanglement and minor degradation of benthic species from line abrasion and the use of weights (sinkers). Schleyer and Tomalin (2000) noted that discarded or lost fishing line appeared to entangle readily on branching and digitate corals and was accompanied by progressive algal growth. This subsequent fouling eventually overgrows and kills the coral, becoming an amorphous lump once accreted by coralline algae (Schleyer and Tomalin 2000). Lines entangled amongst fragile coral may break delicate gorgonians and similar species. Due to the widespread use of weights over coral reef or hardbottom habitat and the concentration of effort over these habitat areas from recreational and commercial fishermen, the cumulative effect may lead to significant impacts resulting from the use of these gear types.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

PATENT TONG

Similar to hand tongs, hydraulic patent tongs (Figure 26) are much larger and are assisted with hydraulic lift, allowing them to purchase more benthic area in pursuit of oysters. Patent tongs are utilized in the oyster fisheries that occur in state waters.

IMPACTS

Rothschild et al. (1994) found that hydraulic-powered patent tongs are the most destructive gear to oyster reef structure because of their capability to penetrate and disassociate the oyster reef. The capability arises from the gear weight and hydraulic power. Patent tongs operate much like an industrial crane with each bite having the ability to remove a section of the oyster bar amounting to $0.25m^3$.

RECOVERY

No information is provided in the literature in regard to recovery metrics. However, it may be noted that recovery may be protracted as fishing intensity increases.

MANAGEMENT RECOMMENDATIONS

Due to overfishing and disease, oysters may now be more economically valuable for the habitat they provide for other valued species than they are for the oyster fishery (Lenihan and Peterson 1998). Rothschild et al. (1994) suggested the establishment of broodstock sanctuaries that includes the designation of “no-fishing” restrictions in specific areas. Lenihan and Micheli (2000) also recommended the closure of some oyster reefs to harvest.

Maintaining high densities of oysters on some intertidal reefs may help to preserve future oyster harvests and broodstock. Furthermore, protecting some reefs will also preserve the ecological functions that oyster reef provide such as improving water quality and providing essential recruitment, refuge, and foraging habitat for numerous marine species.

PURSE SEINE & LAMPARA NET

Purse seines (Figures 27, 28) are walls of netting used to encircle entire schools of fish at or near the surface. Spotter planes are often used to locate the schools, which are subsequently surrounded by the netting and trapped by the use of a pursing or drawstring cable threaded through the bottom of the net. When the cable has pulled the netting tight, enclosing the fish in the net, the net is retrieved to congregate the fish. The catch is then either pumped onboard or hauled onboard with a crane-operated dip net in a process called brailing. Purse seines are utilized to harvest menhaden in the Gulf and South Atlantic. Similarly, the lampara net (Figure 29) has a large central bunt, or bagging portion, and short wings. The buoyed float line is longer than the weighted lead line so that as the lines are hauled the wings of the net come together at the bottom first, trapping the fish. As the net is brought in, the school of fish is worked into the bunt and captured. In the Florida Keys a modified lampara net is used to harvest baitfish near the top of the water column. The wing is used to skim the water surface as the net is drawn in and fish are herded into the pursing section to be harvested with a dip net.

IMPACTS
Purse seines in the Gulf menhaden fishery frequently interact with the bottom, resulting in sediment resuspension.

RECOVERY
Schoellhammer (1996) estimated that sediments resuspended by purse seining activities would last only a period of hours.

MANAGEMENT RECOMMENDATIONS
Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

PUSHNET

Employed to harvest shrimp in shallow water, pushnets (Figure 30) consist of netting supported by a frame that is mounted onto a pole which is then pushed across the bottom. Pushnets are generally utilized on SAV beds where shrimp can be harvested in abundant numbers.

IMPACTS
DeSylva (1954) determined that pushnets have no detrimental effect on habitat.

RECOVERY
Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery are offered.

MANAGEMENT RECOMMENDATIONS
Due to the general lack of impacts and limited nature of this fishery, no management recommendations are offered.

SLURP GUN

A slurp gun (Figure 31) is a self-contained, handheld device that captures tropical fish by rapidly drawing seawater containing such fish into a closed chamber. Slurp guns are typically employed on hardbottom and coral reef habitat in both state and Federal waters.

IMPACTS

It is possible that tropical collectors may impact coral or other benthic invertebrates in pursuit of tropical species that are harvested on hardbottom or coral habitat areas. However, due to the limited force applied by a diver in an errant fin kick or hand placement, the likely effects to habitat would be minor.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

SNARE

Recreational divers pursuing spiny lobster often use a long, thin pole that has a loop of coated wire on the end called a snare (Figure 32). The loop is placed around a lobster that may be residing in a tight overhang or other inaccessible location, and then tightened by a pull toggle at the base of the pole in order to capture and extract the lobster.

IMPACTS

While there are no studies that evaluate this gear type, it is probable that use of this gear may minimize impacts to habitat in comparison to divers that use no additional gear (hand harvest). Due to the more surgical precision with the snare, divers likely impact the surrounding habitat to a lesser extent than if capturing by hand only due to the required leverage needed by the divers to capture a lobster by hand.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

SPEAR & POWERHEAD

Divers use pneumatic or rubber band guns (Figure 33) or slings to hurl a spear shaft to harvest a wide array of fish species. Reef species such as grouper and snapper, as well as pelagic species such as dolphin and mackerel, are targeted by divers. Commercial divers sometimes employ a shotgun shell known as a powerhead at the shaft tip, which efficiently delivers a lethal charge to their quarry. This method is commonly used to harvest large species such as amberjack.

IMPACT

Gomez et al. (1987) concluded that spearfishing on reef habitat may result in some coral breakage, but damage is probably negligible. A study that assessed recreational SCUBA activity in the US Caribbean (Garcia-Moliner et al. 2000) concluded that approximately 0.7% of the total recreational divers in the USVI and 28% of the total recreational divers in Puerto Rico are spearfishing. Potential impact would be approximately 4,736 units in the USVI and 220,264 units in Puerto Rico. In this study, impact units consisted of two hands and two feet (4 units per diver) and impact was broadly defined as ranging from touching coral with hands to the resuspension of sediment by fins. No assessment of habitat degradation or long-term impacts was discussed. It may be assumed that divers pursuing pelagic species have no effect on habitat due to the absence of any interaction with the benthos.

RECOVERY & MANAGEMENT RECOMMENDATIONS

Due to the lack of scientific investigation on potential habitat impacts resulting from this gear, no conclusions on recovery or management recommendations are offered.

CURRENT MANAGEMENT MEASURES TO PROTECT EFH

SOUTH ATLANTIC FISHERY MANAGEMENT COUNCIL

Through the Coral, Coral Reefs, and Live/Hard Bottom Habitat FMP and its subsequent amendments, the South Atlantic Council has protected coral reefs and hardbottom habitat by prohibiting all harvest or possession of these resources, with the exception of a limited fishery for allowable octocorals (species of the subclass Octocorallia, with the exception of *Gorgonia flabellum* and *Gorgonia ventalina*). The designation of the Oculina Bank HAPC prohibited the use of bottom trawls, dredges, pots, traps, or bottom longlines in this fragile habitat area. In its Snapper Grouper FMP, the Council prohibited the use of bottom longlines in the EEZ within 50 fathoms or anywhere south of St. Lucie Inlet, Florida, as well as fish traps, entanglement gear, and bottom trawls on hardbottom habitat. Also under the Snapper Group FMP is an Experimental Oculina Research Reserve where the harvest or possession of all species within the snapper grouper complex is prohibited.

GULF OF MEXICO FISHERY MANAGEMENT COUNCIL

The Gulf of Mexico Council, through its FMPs and amendments to the FMPs, have implemented various regulations that protect and benefit EFH. Seasonal or annual trawl closures, such as the Tortugas Shrimp Sanctuary which protects a considerable area off southwest Florida, have been established through their Stone Crab and Shrimp FMPs. The Reef Fish FMP and its subsequent amendments prohibited the use of poisons and explosives due to their documented impacts on habitat. Gear-specific zones were created which have provided extensive habitat benefits. Fish traps and roller ("rockhopper") trawls were prohibited within an inshore stressed area, following depth contours around the Gulf of between 18.29 - 45.72 meters (60 - 150 feet). Furthermore, longline/buoy gear prohibted areas were established along the 20-fathom contour in the eastern Gulf and the 50-fathom contour in the central-western Gulf. Additionally, two marine reserves which encompass 566.99km^2 (219nm^2) and provide complete protection to habitat and associated marine species, were created off west central Florida to protect gag spawning aggregations. Through the Coral and Coral Reef FMP, the harvest of stony coral, seafans (*Gorgonia flabellum* and *Gorgonia ventalina*), and natural liverock was prohibited and Habitat Areas of Particular Concern (HAPCs) were established off Florida (Florida Middle Ground) and Texas (East and West Flower Garden Bank). These HAPCs are defined by areas dominated with coral species that may easily be degraded by particular fishing activities. Therefore, the use of any fishing gear interfacing with the bottom (i.e., bottom trawls, traps, pots, and bottom longlines) was prohibited within the HAPCs. Amendments to the Coral and Coral Reef FMP also regulated the use of chemicals used by fish collectors near coral reefs.

CARIBBEAN FISHERY MANAGEMENT COUNCIL

Similar to actions initiated by the Gulf of Mexico and South Atlantic Fishery Management Councils, the Caribbean Council prohibited the harvest and possession of corals and live rock through its FMP for Corals and Reef Associated Plants and Invertebrates. A recent amendment to the FMP established the Hind Bank Marine Conservation District (MCD) off St. Thomas, U.S. Virgin Islands. Within this MCD, fishing for any species is prohibited. The creation of this marine protected area provides complete protection to the local marine ecosystem under the Magnuson-Stevens Act.

SUMMARY

Habitat is constantly degraded by a variety and combination of negative impacts such as bioturbation, pollution, storm events, coastal development, and fishery-related impacts. While pollution and development may present a far more insidious threat, fishery-related impacts represent a direct potential threat to EFH and must be evaluated under the Magnuson-Stevens Act. Reviewing the information provided in this study indicates that several fishing activities have negligible or minor impacts on EFH. As these conclusions are based on available information, it is feasible that other, undocumented impacts may occur during fishing activities. Additionally, the

absence of long-term studies and a lack of control sites hinder the ability to properly evaluate cumulative impacts. Therefore, caution should be exercised in declaring the impacts of particular fishing activities minor or negligible.

Trawling activities have come under close scrutiny due to numerous claims of widespread habitat destruction. Comparisons to forest clear-cutting have been offered in other studies (Watling and Norse 1998). However, given the available scientific information, it would appear that trawling has a minor physical impact to EFH in many areas of the Gulf of Mexico and South Atlantic. Trawls harvesting shrimp frequently operate over sandy or muddy habitat areas. The major result of these activities would be sediment resuspension which is a relatively minor and short-term impact. It should be noted that increased sedimentation may have more serious biological consequences in estuarine areas where variances in nutrient cycling may dramatically affect the localized ecosystem. Furthermore, sediment resuspension may have serious consequences in areas where heavy metals and other contaminants are found.

Special consideration should be taken when evaluating complex benthic habitat such as coral reefs. Fishing in general is a potential threat to the sustainability of coral reef habitats; due to the interspecies relationships within a coral reef community, targeting and extraction of a particular species may disturb the system and subject the reef to other stressors (Dayton et al. 1995; Jennings and Polunin 1996). Sponges and corals represent the largest and most conspicuous sessile species in hardbottom habitats in the South Atlantic (Van Dolah et al. 1987). The entire demersal stage of the life histories of many species associated with coral reefs have obligate habitat requirements or demonstrate recruitment bottlenecks. Without the specific structural components of habitat, the populations of fishes with these habitat requirements would not persist (Auster and Langton 1999). The degradation of hardbottom communities and coral reefs may reduce the amount of habitat for other species. Since competition can occur for space as well as for food (Paine 1974), fishing impacts may introduce additional stress to reef associated species, as well as to the habitat.

Oyster reefs also warrant special consideration. Impacts to oyster reefs, especially fishing activities that target oysters, directly reduce EFH and hamper the natural water-cleansing ability of oysters (Coen 1995). Furthermore, fishing activities adjacent to oyster reefs can have a significant impact. The oyster fishery in the Chesapeake Bay is perhaps the best example of the ramifications of habitat degradation. Rothschild et al. (1994) contended that fishing, both the removal of oyster and the associated degradation of oyster reef habitat, may be more important to the decline of oysters in Chesapeake Bay than either water quality or disease. The removal of any reef-building species, such as oysters, will inevitably result in large changes in the species assemblages associated with the reef structure itself (ICES 1995).

As previously mentioned, the empirical study of fishing effects is hampered by a lack of unfished control sites (Dayton et al. 1995; Jennings and Kaiser 1998). To quantify the effects of disturbance, one must use an experimental approach that compares fished (e.g., by trawls) and unfished sites (Van Dolah 1987; Collie et al. 1997). Additionally, one of the greatest challenges in assessing the effects of fishing on habitat is the lack of knowledge of potential for recovery, succession, and resilience to fishing activities (Cappo et al. 1998). Little has been written about the recovery of seafloor habitat from fishing gear effects. The are few, if any, areas within the Region that provide the opportunities to evaluate fishing impacts on "natural" habitat areas. It should be noted that "no-take" zones, gear zoning, or area rotation depending on particular gear and habitat type is the most prevalent management recommendation in the reviewed literature (Gomez et al. 1987; ICES 1991; ICES 1992; Van der Knapp 1993; McAllister and Spiller 1994; Rothschild et al. 1994; ICES 1995; Sargent et al. 1995; Auster et al. 1996; Macdonald et al. 1996; Sainsbury et al. 1997; Collie 1998; Engel and Kvitek 1998; Goñi 1998; Hall 1999; Jennings and Kaiser 1998; Lindeboom and de Groot 1998; Watling and Norse 1998; Friedlander et al. 1999; Turner et al. 1999; Bergman and Santbrink 2000; Kaiser 2000). This management recommendation may not only provide adequate and prudent habitat protection, but also the ability to better evaluate the impacts of fishing.

In many cases, fishery-related impacts may occur due to the lack of knowledge that there is a potential for an impact. The lack of detailed mapping and accurate habitat designations prevents the protection of many areas. Perhaps, one of the most beneficial exercises in an attempt to prevent fishery-related impacts would be the precise mapping of habitat. This review illustrates that several gear types are not compatible with certain habitat types (e.g., otter trawl working hardbottom and coral reefs). Once sufficient habitat maps are available, it would be possible to designate appropriate gear restrictions which, in turn, may effectively prevent further fishery-related impacts to the extent practicable.

While this review attempts to improve the knowledge base of fishery-related impacts within the Southeast Region, it is by no means complete nor entirely conclusive. As noted by Taylor (1956), "calm discussion based on scientific research should discover the answers. The pure scientist possibly could not reach a satisfactory conclusion under a lifetime of study. Then, he might not be satisfied that all knowledge of the subject had been gained. For day to day living, often it is necessary to proceed without all the facts. It may be required that certain assumptions be adopted as a guide. It should be sufficient that these assumptions are based upon clear knowledge of the basic facts. Let it be certain that these basics are facts, however - not assumptions." This observation, made 45 years earlier, accurately reflects the current situation managers are confronted with in regard to fishery-related habitat impacts.

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DEFINITIONS

([1]Florida Fish and Wildlife Conservation Commission - Division of Marine Fisheries, [2] Texas Parks and Wildlife, [3]Louisiana Department of Wildlife and Fisheries, [4]Code of Federal Regulations)

Allowable chemical: means a substance, generally used to immobilize marine life so that it can be captured alive, that, when introduced into the water, does not take Gulf and South Atlantic prohibited coral and is allowed by Florida for the harvest of tropical fish (e.g., quinaldine, quinaldine compounds, or similar substances).[4]

Artificial lure: any lure (including flies) with hook or hooks attached that is man-made and is used as a bait while fishing.[2]

Automatic reel: means a reel that remains attached to a vessel when in use from which a line and attached hook(s) are deployed. The line is payed out from and retrieved on the reel electrically or hydraulically.[4]

Bait: something used to lure any wildlife resource.[2]

Beach or haul seine: means a seine that is hauled or dragged over the bottom into shallow water or onto the beach, either by hand or with power winches.[1]

Bully net: means a circular frame attached at right angles to the end of a pole and supporting a conical bag of webbing. The webbing is usually held up by means of a cord which is released when the net is dropped over a lobster.[4]

Buoy gear: means fishing gear consisting of a float and one or more weighted lines suspended therefrom, generally long enough to reach the bottom. A hook or hooks (usually 6 to 10) are on the lines at or near the end. The float and line(s) drift freely and are retrieved periodically to remove catch and rebait hooks.[4]

Butterfly net: a fixed, frame-mounted net, used to fish near-surface waters, which is suspended from the side or sides of a boat, pilings, floats, rafts or shore installation.[3]

Cast net: means a cone-shaped net thrown by hand and designed to spread out and capture fish as the weighted circumference sinks to the bottom and comes together when pulled by a line.[1]

Crab dropnet: any device constructed with vegetable, synthetic, or metal fibers and without flues or throat, attached to a wire frame that forms a net basket and is used for the purpose of taking crabs. This device shall be operated solely by hand and fished in a stationary, passive manner.[3]

Crab trap: a cube-shaped device with entrance funnels and either a bait box or materials providing cover or shelter for peeler crabs, which is used for the sole purpose of taking crabs. This device shall be fished in a stationary, passive manner.[3]

Dip net: a net, usually a deep mesh bag of vegetable or synthetic materials, on a fixed frame attached to a handle and held and worked exclusively by hand and by no more than one individual. *see also Landing net.*[3]

Drift gillnet: means a gillnet, other than a long gillnet or a run-around gillnet, that is unattached to the ocean bottom, regardless of whether attached to a vessel.[4]

Entangling net: means a drift net, trammel net, stab net, or any other net which captures saltwater finfish,

shellfish, or other marine animals by causing all or parts of heads, fins, legs, or other body parts to become entangled or ensnared in the meshes or in the pockets of the net. This term does not include a cast net.[1]

Fish trap: (2) In the Gulf EEZ, a trap and its component parts (including the lines and buoys), regardless of the construction material, used for or capable of taking finfish, except a trap historically used in the directed fishery for crustaceans (that is, blue crab, stone crab, and spiny lobster).[4]

Fold-up trap: a device utilized to capture crabs which is baited and lowered to the bottom. When recovered, side panels fold up to capture crabs on the base panel.*

Fyke net: any cone-shaped net of vegetable or synthetic fibers having throats or flues which are stretched over a series of rings or hoops to support the webbing, with vertical panels of net wings set obliquely on one or both sides of the mouth of the cone-shaped net.[3]

Gaff: any hand held pole with a hook attached directly to the pole.[2]

Gig: any hand held shaft with single or multiple points, barbed or barbless.[2]

Gill net: means one or more walls of netting which captures fish by ensnaring or entangling them in the meshes of the net by the gills. This term does not include a cast net.[1]

Handline: means a line with attached hook(s) that is tended directly by hand.[4]

Hook and line gear: means any handline, rod, reel, or any pole to which hook and line are attached, as well as any bob, float, weight, lure, plug, spoon, or standard bait attached thereto, with a total of ten or fewer hooks.[1]

Hoop net: 1. a cone-shaped net of vegetable or synthetic materials having throats or flues and which are stretched over a series of rings or hoops to support the webbing.[3] 2. A frame, circular or otherwise, supporting a shallow bag of webbing and suspended by a line and bridles. The net is baited and lowered to the ocean bottom, to be raised rapidly at a later time to prevent the escape of lobster.[4]

Landing or dip net: means a hand-held net consisting of a mesh bag suspended from a circular, oval, or rectangular rigid frame attached to a handle.[1]

Lead or wing net: a panel of netting of any mesh size or length, with or without weights and floats, attached to one or both sides of the mouth of a cone-shaped net having flues or throats, and set so as to deflect or guide fish toward the mouth of the net.[3]

Long gillnet: means a gillnet that has a float line that is more than 1,000 yd (914 m) in length.[4]

Longline: means a line that is deployed horizontally to which gangions and hooks are attached. A longline may be a bottom longline, i.e., designed for use on the bottom, or a pelagic longline, i.e., designed for use off the bottom. The longline hauler may be manually, electrically, or hydraulically operated.[4]

Menhaden seine: a purse seine used to take menhaden and herring-like species.[3]

Mesh area (of a net): means the total area of netting with the meshes open to comprise the maximum square footage. The square footage shall be calculated using standard mathematical formulas for geometric shapes. The square footage of seines and other rectangular nets shall be calculated using the maximum length and maximum width of the netting.[1]

Mesh size: the full measure of the mesh as found in use when measured as follows: Bar measure is the length of the full bar stretched from the near side of one knot to the far side of the other after being tarred, treated, or otherwise processed. Stretched measure is the full stretched distance from the near side of one knot to the far side of the opposite knot diagonally across the mesh. This measurement shall not be applicable to weaved or woven nets commonly used for menhaden fishing. In woven nets, stretched measure is the full stretched distance of the opening of the mesh; bar measure is one-half of stretched measure.[3]

Monofilament: a single untwisted synthetic filament.[3]

Mullet strike net: a gill net that is not more than 1,200 feet long and with a mesh size of not less than 3 ½ inches stretched that is not anchored or secured to the water bottom or shore and which is actively worked while being used.[3]

Multiple hook: means two or more fishhooks bound together to comprise a single unit or any hook with a single shank and eye and two or more pointed ends, used to impale fish.[1]

Pompano strike net: a gill net that is not more than 2,400 feet long and with a mesh size of not less than 5 inches stretched that is not anchored or secured to the water bottom or shore and which is actively worked while being used.[3]

Powerhead: means any device employing an explosive charge or a release of compressed gas, usually attached to a speargun, spear, pole, or stick (known as a "bangstick"), which detonates upon contact.[1]

Purse seine: any net or device commonly known as a purse seine and/or ring net that can be pursed or closed by means of a drawstring or other device that can be drawn to close the bottom of the net or the top of the net or both. Such nets are constructed of mesh of such size and design as not to be used primarily to entangle fish by the gills or other bony projection.[3]

Rebreather: means a closed circuit or semi-closed circuit underwater breathing apparatus that recycles and recirculates all or part of the gas mixture supplied for breathing. A rebreather is distinguished from other underwater breathing apparatuses by the inclusion of a scrubber (a component that removes carbon dioxide from the breathing gas) and a counterlung (a waterproof bag that allows the diver's exhaled breath to be captured for scrubbing and recycling back to the diver for inhalation).[1]

Rod and reel: means a rod and reel unit that is not attached to a vessel, or, if attached, is readily removable, from which a line and attached hook(s) are deployed. The line is payed out from and retrieved on the reel manually, electrically, or hydraulically.[4]

Run-around gillnet: means a gillnet, other than a long gillnet, that, when used, encloses an area of water.[4]

Sail Line: type of trotline with one end of the main line fixed on the shore, the other end of the main line attached to a wind-powered floating device or sail.[2]

Sea bass pot: means a trap has six rectangular sides and does not exceed 25 inches (63.5 cm) in height, width, or depth.[4]

Seine: means a small-meshed net suspended vertically in the water, with floats along the top margin and weights along the bottom margin, which encloses and concentrates fish, and does not entangle them in the meshes.[1] *see also Purse seine.*

Skimmer net: a net attached on two sides to a triangular frame and suspended from or attached to the sides of a boat, with one corner attached to the side of the boat and one corner resting on the waterbottom. A ski and one end of the lead line are attached to the corner of the frame that rests on the waterbottom and the other end of the lead line is attached to a weight which is suspended from the bow of the boat.[3]

Spear: any shaft with single or multiple points, barbed or barbless, which may be propelled by any means, but does not include arrows.[2]

Speargun: any hand operated device designed and used for propelling a spear, but does not include the crossbow.[2]

Stab or sink net: means a gill or trammel net, that sinks to the bottom when placed, set, or fished in water deeper than its hanging depth.[1]

Strike net: any gill net, trammel net or seine not anchored or secured to the water bottom or shore and which is actively worked while being used.[3]

Test trawl: a trawl which is not more than 16 feet along the corkline or 20 feet along the leadline or headrope.[3]

Trammel net: means a net constructed of two or more walls of netting hung from the same cork and lead lines, with one wall having a larger mesh than the other(s), which traps a fish in a pocket of netting when the fish pushes the smaller mesh wall through a mesh in the larger mesh wall.[1]

Trawl: any net, generally funnel-shaped, pulled through the water or along the bottom with otter boards to spread the mouth open while being fished. The term "trawl" also means and includes plumb staff beam trawls that do not exceed 16 feet, and that do not use otter boards but are held open laterally by a horizontal beam and vertically by two vertical beams (plumb staffs), and that are used while the vessel is under way.[3]

Trawl (Individual Bait-Shrimp Trawl): a bag-shaped net which is dragged along the bottom or through the water to catch aquatic life.[2]

Trotline: a non-metallic main fishing line with more than five hooks attached and with each end attached to a fixture.[2]

Umbrella net: a non-metallic mesh net that is suspended horizontally in the water by multiple lines attached to a rigid frame.[2]

Underwater breathing apparatus: means any apparatus, whether self-contained or connected to a distant source of air or other gas, whereby a person wholly or partially submerged in water is able to obtain or reuse air or any other gas or gasses for breathing without returning to the surface of the water.[1]

Wing (with reference to a seine): means a panel of netting on one or both ends of the seine, which panel has a larger mesh than the main body of the seine and is used to guide fish into the main body of the seine.[1]

APPENDIX: LIST OF AUTHORIZED GEAR (64 FR 67511)

SOUTH ATLANTIC FISHERY MANAGEMENT COUNCIL

Golden Crab Fishery (FMP)	Trap.
Crab Fishery (Non-FMP):	
A. Dredge fishery	A. Dredge.
B. Trawl fishery	B. Trawl.
C. Trap and pot fishery	C. Trap, pot.
Atlantic Red Drum Fishery (FMP)	No harvest or possession in the EEZ.
Coral and Coral Reef Fishery (FMP):	
A. Octocoral commercial fishery	Hand harvest.
B. Live rock aquaculture fishery	Hand harvest.
South Atlantic Shrimp Fishery (FMP)	Trawl.
South Atlantic Snapper-Grouper Fishery (FMP):	
A. Commercial fishery	A. Longline, rod and reel, bandit gear, handline, spear, powerhead.
B. Black sea bass trap and pot fishery	B. Pot, trap.
C. Wreckfish fishery	C. Rod and reel, bandit gear, handline.
D. Recreational fishery	D. Handline, rod and reel, bandit gear, spear, powerhead.
South Atlantic Spiny Lobster Fishery (FMP):	
A. Commercial fishery	A. Trap, pot, dip net, bully net, snare, hand harvest.
B. Recreational fishery	B. Trap, pot, dip net, bully net, snare, hand harvest.
South Atlantic Coastal Migratory Pelagics Fishery (FMP):	
A. Commercial Spanish mackerel fishery	A. Handline, rod and reel, bandit gear, gillnet, cast net.
B. Commercial king mackerel fishery	B. Handline, rod and reel, bandit gear.
C. Other commercial coastal migratory pelagics fishery	C. Longline, handline, rod and reel, bandit gear
D. Recreational fishery	D. Bandit gear, rod and reel, handline, spear.
Spiny Dogfish Fishery (FMP jointly managed by NEFMC and SAFMC):	
A. Gillnet fishery	A. Gillnet.
B. Trawl fishery	B. Trawl.
C. Hook and line fishery	C. Hook and line, rod and reel, spear, bandit gear.
D. Dredge fishery	D. Dredge.
E. Longline fishery	E. Longline.
F. Recreational fishery	F. Hook and line, rod and reel, spear.
Smooth Dogfish Fishery (Non-FMP):	
A. Gillnet fishery	A. Gillnet.
B. Trawl fishery	B. Trawl.
C. Hook and line fishery	C. Hook and line, rod and reel, spear, bandit gear.
D. Dredge fishery	D. Dredge.
E. Longline fishery	E. Longline.
F. Recreational fishery	F. Hook and line, rod and reel, spear.
Atlantic Menhaden Fishery (Non-FMP):	
A. Purse seine fishery	A. Purse seine.
B. Trawl fishery	B. Trawl.
C. Gillnet fishery	C. Gillnet.
D. Commercial hook-and-line	D. Hook and line fishery.
E. Recreational fishery	E. Hook and line, snagging, cast nets.
Atlantic Mackerel, Squid, and Butterfish Trawl Fishery (Non-FMP)	Trawl.
Bait Fisheries (Non-FMP)	Purse seine.
Weakfish Fishery (Non-FMP):	
A. Commercial fishery	A. Trawl, gillnet, hook and line.
B. Recreational fishery	B. Hook and line, spear.
Whelk Fishery (Non-FMP):	
A. Trawl fishery	A. Trawl.
B. Pot and trap fishery	B. Pot, trap.
C. Dredge fishery	C. Dredge.
D. Recreational fishery	D. Hand harvest.
Marine Life Aquarium Fishery (Non-FMP)	Dip net, slurp gun, barrier net, drop net, allowable chemical, trap, pot, trawl.

Calico Scallop Fishery (Non-FMP):	
A. Dredge fishery	A. Dredge.
B. Trawl fishery	B. Trawl.
C. Recreational fishery	C. Hand harvest.
Summer Flounder Fishery (FMP managed by MAFMC):	
A. Commercial fishery	A. Trawl, longline, handline, rod and reel, pot, trap, gillnet, dredge.
B. Recreational fishery	B. Rod and reel, handline, pot, trap, spear.
Bluefish, Croaker, and Flounder Trawl and Gillnet Fishery (Bluefish FMP managed by MAFMC)	Trawl, gillnet.
Commercial Fishery (Non-FMP)	Trawl, gillnet, longline, handline, hook and line, rod and reel, bandit gear, cast net, pot, trap, lampara net, spear.
Recreational Fishery (Non-FMP)	Rod and reel, handline, spear, hook and line, hand harvest, bandit gear, powerhead, gillnet, cast net.
Sargassum Fishery (Non-FMP)	Trawl.
Octopus Fishery (Non-FMP)	Trap, pot.

GULF OF MEXICO FISHERY MANAGEMENT COUNCIL

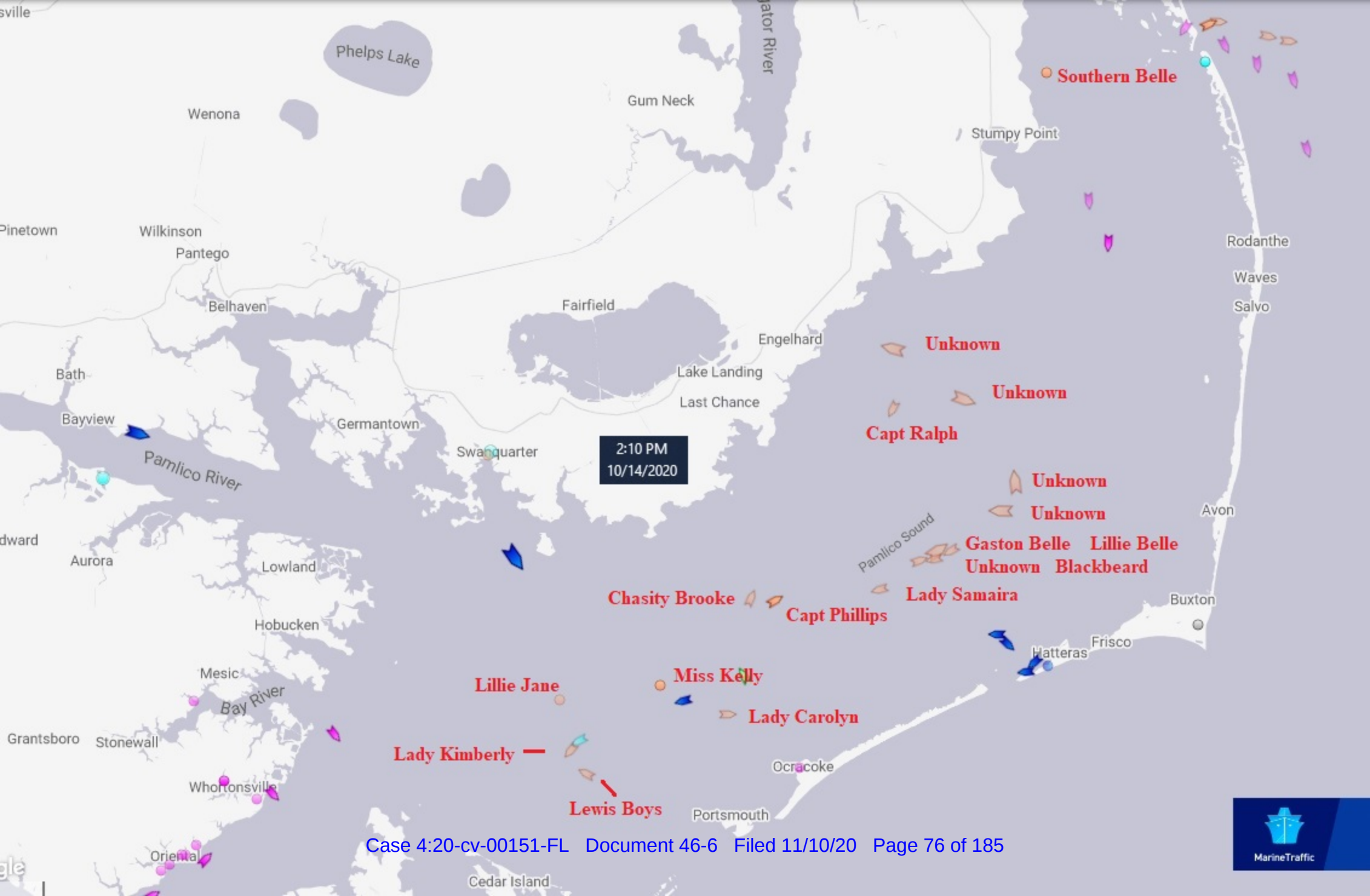
Gulf of Mexico Red Drum Fishery (FMP)	No harvest or possession in the EEZ.
Coral Reef Fishery (FMP):	
A. Commercial fishery	A. Hand harvest.
B. Recreational fishery	B. Hand harvest.
Gulf of Mexico Reef Fish Fishery (FMP):	
A. Snapper-Grouper reef fish longline and hook and line fishery.	A. Longline, handline, bandit gear, rod and reel, buoy gear.
B. Pot and trap reef fish fishery	B. Pot, trap.
C. Other commercial fishery	C. Spear, powerhead, cast net, trawl.
D. Recreational fishery	D. Spear, powerhead, bandit gear, handline, rod reel, cast net.
Gulf of Mexico Shrimp Fishery (FMP):	
A. Gulf of Mexico commercial fishery	A. Trawl butterfly net, skimmer, cast net.
B. Recreational fishery	B. Trawl.
Gulf of Mexico Coastal Migratory Pelagics Fishery (FMP):	
A. Large pelagics longline fishery	A. Longline.
B. King/Spanish mackerel gillnet fishery	B. Gillnet.
C. Pelagic hook and line fishery	C. Bandit gear, handline, rod and reel.
D. Pelagic species purse seine fishery	D. Purse seine.
E. Recreational fishery	E. Bandit gear, handline, rod and reel, spear.
Gulf of Mexico Spiny Lobster Fishery (FMP):	
A. Commercial fishery	A. Trap, pot, dip net, bully net, hoop net, trawl, snare, hand harvest.
B. Recreational fishery	B. Dip net, bully net, pot, trap, snare, hand harvest.
Stone Crab Fishery (FMP):	
A. Trap and pot fishery	A. Trap, pot
B. Recreational fishery	B. Trap, pot, hand harvest.
Blue Crab Fishery (Non-FMP)	Trap, pot.
Golden Crab Fishery (Non-FMP)	Trap.
Mullet Fishery (Non-FMP):	
A. Trawl fishery	A. Trawl.
B. Gillnet fishery	B. Gillnet.
C. Pair trawl fishery	C. Pair trawl.
D. Cast net fishery	D. Cast net.
E. Recreational fishery	E. Bandit gear, handline, rod and reel, spear, cast net.
Inshore Coastal Gillnet Fishery (Non-FMP)	Gillnet.
Octopus Fishery (Non-FMP)	Trap, pot.
Marine Life Aquarium Fishery (Non-FMP)	Dip net, slurp gun, barrier net, drop net, allowable chemical, trap, pot, trawl.
Coastal Herring Trawl Fishery (Non-FMP)	Trawl.
Butterfish Trawl Fishery (Non-FMP)	Trawl.
Gulf of Mexico Groundfish (Non-FMP):	
A. Commercial fishery	A. Trawl, purse seine, gillnet.

B. Recreational fishery	B. Hook and line, rod and reel, spear.
Gulf of Mexico Menhaden Purse Seine Fishery (Non-FMP)	Purse seine.
Sardine Purse Seine Fishery (Non-FMP)	Purse seine.
Oyster Fishery (Non-FMP)	Dredge, tongs.
Commercial Fishery (Non-FMP)	Trawl, gillnet, hook and line, longline, handline, rod and reel, bandit gear, cast net, lampara net, spear.
Recreational Fishery (Non-FMP)	Bandit gear, handline, rod and reel, spear, bully net, gillnet, dip net, longline, powerhead, seine, slurp gun, trap, trawl, harpoon, cast net, hoop net, hook and line, hand harvest.

CARIBBEAN FISHERY MANAGEMENT COUNCIL

Caribbean Spiny Lobster Fishery (FMP):	
A. Trap/pot fishery	A. Trap/pot.
B. Dip net fishery	B. Dip net.
C. Entangling net fishery	C. Gillnet, trammel net.
D. Hand harvest fishery	D. Hand harvest, snare.
E. Recreational fishery	E. Dip net, trap, pot, gillnet, trammel net.
Caribbean Shallow Water Reef Fish Fishery (FMP):	
A. Longline/hook and line fishery	A. Longline, hook and line.
B. Trap/pot fishery	B. Trap, pot.
C. Entangling net fishery	C. Gillnet, trammel net.
D. Recreational fishery	D. Dip net, handline, rod and reel, slurp gun, spear.
Coral and Reef Resources Fishery (FMP):	
A. Commercial fishery	A. Dip net, slurp gun.
B. Recreational fishery	B. Dip net, slurp gun, hand harvest.
Queen Conch Fishery (FMP):	
A. Commercial fishery	A. Hand harvest.
B. Recreational fishery	B. Hand harvest.
Caribbean Pelagics Fishery (Non-FMP):	
A. Pelagics drift gillnet fishery	A. Gillnet.
B. Pelagics longline/hook and line fishery	B. Longline/hook and line.
C. Recreational fishery	C. Spear, handline, longline, rod and reel.
Commercial Fishery (Non-FMP)	Trawl, gillnet, hook and line, longline, handline, rod and reel, bandit gear, cast net, spear.
Recreational Fishery (Non-FMP)	Rod and reel, hook and line, spear, powerhead, handline, hand harvest, cast net.

Exhibit B


Phelps Lake
Wenona
Gum Neck
Stumpy Point
Southern Belle
Pinetown
Wilkinson
Pantego
Belhaven
Fairfield
Engelhard
Lake Landing
Last Chance
Rodanthe
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Blackbeard
Aurora
Lowland
Chasity Brooke
Capt Phillips
Lady Samaira
Buxton
Hobucken
Frisco
Hatteras
Mesic
Bay River
Lillie Jane
Miss Kelly
Lady Carolyn
Grantsboro
Stonewall
Lady Kimberly
Ocracoke
Whortonsville
Lewis Boys
Portsmouth
Oriental
Cedar Island
MarineTraffic


LAND VIEWER
Locations, coordinates, scene IDs
Upload area of interest (AOI)
14 Oct 2020
Sentinel-2 L2A
44°
0%
18SVE
4 out of 10 free scenes per day
Upgrade
1
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LAND VIEWER
Locations, coordinates, scene IDs
Upload area of interest (AOI)
14 Oct 2020
Sentinel-2 L2A
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14 Oct 2020
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LAND VIEWER
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Exhibit C

Environ Syst Decis (2015) 35:334–350
DOI 10.1007/s10669-015-9557-2

REVIEW

A review of the potential effects of suspended sediment on fishes: potential dredging-related physiological, behavioral, and transgenerational implications

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Abstract The long-term effects of sediment exposure on aquatic organisms are poorly understood, yet it is critical for determining threshold effects and exposure limits to mitigate potential impacts with regard to population dynamics. In this paper, we present the current state of knowledge to help consolidate the breadth of information regarding total suspended solids (TSS) thresholds for aquatic species, as well as identify areas where data are lacking. More specifically, we provide the state of the science related to TSS effects on freshwater and estuarine fish including short-term (i.e., physiology and behavior) and long-term effects. Our research indicated that little attention has been given to examining long-term effects, e.g., transgenerational effects, from suspended sediments (SS) on fish populations. Understanding transgenerational effects is paramount to developing and predicting the links between fish condition, survival, populations, and communities. Survival of a local fish population to high sediment loads often translates into short-term physiological and behavioral effects; however, the ramifications of such exposure events are rarely tracked across generations. The majority of studies involving SS effects on fish have focused on exposure and mortality rates of affected fish, deposited eggs, or larvae. We developed a conceptual model that highlighted the interactions between sediment dynamics and fish populations. The model can assist in the formulation of more quantitative-based approaches for modeling these interactions. Future research efforts should focus on developing an understanding of whether environmental disturbances, e.g., dredging, may lead to epigenetic changes that may lead to cascade population effects, and if so, under what circumstances.

Keywords Suspended sediments · Dredging · Physiology · Behavior · Population dynamics · Epigenetics

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1 Introduction

Modern commerce relies on navigable waterways to maintain commodity transport across the globe. For example, in 2013, 3,165 tons of goods were transported throughout the U.S. waterways (U.S. Water System 2015). River channel maintenance relies on frequent dredging to keep the waterways navigable. The U.S. Army Corps of Engineers (USACE) dredges, on average, over 147 million cubic yards of sediment annually. Some of the material is removed from the system and used beneficially for beach and wetland nourishment, or habitat creation, while other material may be re-deposited back into the system. Dredging is a complex activity, and its impact on aquatic ecosystems is poorly understood, particularly over long timescales. For example, species are exposed to dredge-created suspended sediment plumes if they are in close proximity to the dredge. However, these plumes are ephemeral and vary in concentration, lessening the further away from the source. Likewise, if the species is vagile, it can choose to move away from the sediment source once exposed. Further, the sensitivity of a species can change ontogenetically, so impacts can be mitigated if a less sensitive life stage is exposed, (e.g., the fry life stage vs an adult). Most of the research in this arena has focused on

impacts of sediment exposure on aquatic species. Notably, there have been few studies which focused exclusively on the relationship between dredging activities and population dynamics of aquatic species. In this paper, we review the current state of knowledge on sediment–fish interactions and attempt to place the work in the context of developing research tools and methods to help begin to understand the impacts of dredging on fish communities.

Sediments are particles that have been suspended, transported, and deposited by water, and are natural and important to elemental cycling in rivers, lakes, and coastal ecosystems (Nichols 1999; Beussink 2007). Suspended sediment events are important, nationally and internationally, as nutrients and contaminants fluxes occur in lakes, rivers, estuaries, coastal systems, and oceans. The rate and magnitude of SS events can be altered by factors such as changes in land use and anthropogenic activities. Suspended solids are often of two general types: inorganic and organic. Inorganic sediments can be described based on mineral composition, origin, particle shape, size, and distribution (Nichols 1999; Beussink 2007), whereas the organic component is biodegradable and varies with origin, amount, and stage of particulate organic matter (Wood 1997). Sediment transport is a function of water movement, and sediment characteristics and composition resulting in a variety of transportation modes. Bed load and suspended load are the primary transportation modes of sediment. Bed load describes sediment particles that move along the bed. Suspended load refers to the suspension of small particles (typically clays and silts <62 μm in diameter) that are carried in the water column (Waters 1995; Garcia 2008). Increases in suspended sediments loads, frequencies, and timing of events are often related directly to anthropogenic activities (e.g., vessels, navigation maintenance and construction, port and road construction, mining, agriculture, logging, and urban development), and indirectly through altered precipitation patterns, increased temperatures, and changes in hard freezes, snowpacks, and snow melts related to changing climates.

Suspended sediment concentrations or water clarity can be directly measured as total suspended sediments (TSS) in mg/L, but more frequently are indirectly measured as turbidity (Fleming et al. 2005). Turbidity is a parameter that is an expression of the optical properties in a sample, and is a measure of the light rays being scattered and absorbed rather than transmitted in straight lines through the sample. Because turbidity measurements are influenced by other compounds and organisms, the correlative relationship of TSS to nephelometric turbidity units (NTUs) is, at best, temporally and spatially explicit.

Suspended sediments can elicit a short- and long-term response from aquatic biota depending on the quantity, quality, and duration of suspended sediment exposure (Caux et al. 1997; Newcombe 2003; Fleming et al. 2005). In 1998, about 40 % of assessed river miles in the U.S.A. had sediment stress-related issues (U.S. EPA 2000). Suspended and bedded sediment (SABS) loading imbalance in aquatic systems can be considered one of the greatest causes of impaired water quality (U.S. EPA 2003; Berry et al. 2003).

In the U.S.A., a universal measurement for SABS does not exist, nor do standard durations for SABS effects testing (Berry et al. 2003). The U.S. EPA has set the following recommendations for developing a numeric criterion for suspended solids and turbidity, i.e., "Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 % from the seasonally established norm for aquatic life" (U.S. EPA 2006). However, there currently exists a wide range of turbidity criteria utilized in the U.S.A. (Berry et al. 2003). The criteria used for SABS can be numerical, narrative, a combination of both, or none at all. The U.S. EPA conducted a study of published SABS criteria in all states in 2001. Numeric SABS criteria existed in 32 of the 53 states, tribes, and territories, and the District of Columbia (U.S. EPA 2006), of which only 30 had criteria for turbidity and seven for suspended solids and five listed criteria for both turbidity and suspended solids. Criteria were in the form of exceedances over background (e.g., "not more than 10 % above background" or "no more than 10 NTUs above background") or absolute values (e.g., "not greater than 100 NTU") (U.S. EPA 2006). Several states provide criteria for an averaging period (e.g., 30 days) as well as an allowed daily maximum concentration (Berry et al. 2003). Other states use exceedances over background (e.g., "not greater than 50 NTU over background", or "not more than 10 % above background"), while some use absolute values (e.g., "not greater than 100 NTU"). There are not many states that use suspended solids as a water quality criterion and values vary from 30 up to 158 mg/L (Berry et al. 2003). Importantly, both the duration (Newcombe and MacDonald 1991) and frequency (Shaw and Richardson 2001) of SAB exposures should be considered when establishing guidelines for exposure thresholds for aquatic organisms (Berry et al. 2003).

In comparison, the regulatory criteria set forth by British Columbia for the protection of aquatic life are as follows: "(1) For clear flow periods, induced turbidity should not exceed background levels by more than 8 NTU during any 24-h period (hourly sampling preferred). For sediment inputs that last between 24 h and 30 days (daily sampling preferred), the mean turbidity should not exceed background by more than 2 NTU. (2) For turbid flow periods, induced turbidity should not exceed the background levels by more than 8 NTU at any time when background turbidity is between 8 and 80 NTU. When background

exceeds 80 NTU, turbidity should not be increased by more than 10 % of the measured background level at any one time. (3) The clear and turbid flow periods are defined by the portion of the hydrograph when suspended sediment concentrations are low (taken to be less than 8 NTU) and relatively elevated (taken to be greater than or equal to 8 NTU), respectively" (Fleming et al. 2005).

Suspended sediments absorb heat energy thereby raising water temperatures (Ellis 1936; Reid 1961; Ryder and Pesendorfer 1989). Turbidity can reduce light transmission through the water and decrease photosynthesis by aquatic plants, consequently affecting dissolved oxygen levels (Berry et al. 2003). As noted in Coen (1995), such effects of turbidity on water quality may result in biological effects on aquatic organisms such as disruptions in migrations and spawning, movement patterns, sublethal effects (e.g., disease susceptibility, growth, and development), reduced hatching success, and direct mortality. Effects of suspended sediments on fish depend upon several factors: species, temperature at exposure time (e.g., Servizi and Martens 1991), type of suspended sediment, [i.e., particle size (Muck 2010) and angularity (e.g., Lake and Hinch 1999)] sediment contaminants (Matta et al. 1999), duration and frequency of exposure, and dose.

In this paper, we present a review of the existing literature in order to help identify the current scope of information available regarding total suspended solids thresholds for fish species. We then develop a conceptual model of the current understanding of the relationship between sediment dynamics and exposure, and fish species. We also identify areas where further research is necessary. The goal of this review is to provide the state of the science related to suspended sediments effects on freshwater and estuarine fish including short-term (i.e., physiology and behavior) and long-term effects (transgenerational).

2 Methods

We began our investigation using a Boolean search using Google Scholar (scholar.google.com). Given the immense amount of literature related to general search terms as "dredging and fish," we explicitly focused the search to a combination of terms: dredging, fish, physiology, behavior, and epigenetic inheritance or transgenerational.

3 Results

The Google Scholar search engine retrieved approximately 61,000 results related to more general search terms as "dredging and fish." The search terms of "dredging and fish and behavior" or "dredging and fish and physiology" produced about 32,800 and 19,500 citations, respectively. "Dredging and fish gametes" produced 3,050 citations, a reduction of about 95.3 %. Interestingly, the search phrase "dredging and fish and transgenerational" or "dredging effects and fish gametes and transgenerational" produced only 136 and 14 results, respectively, a reduction of about 99.7 and 99.9 %. The aforementioned results demonstrate the little attention given to examining long-term, insidious effects, such as transgenerational effects, from suspended sediments on fish populations. Understanding transgenerational effects is paramount to developing and predicting the links between fish condition, survival, populations, and communities. For example, survival of a local fish population to high sediment loads often translates into short-term physiological and behavioral effects; however, often full ramifications of such effects are not tracked into the next generation. Presently, the majority of the studies involving suspended sediment effects on fish have focused on exposure and mortality rates of the exposed fish or deposited eggs and larvae. Accordingly, a review was conducted of more than 150 peer-reviewed papers and reports pertaining to suspended sediments, dredging, and potential effects on fish and other aquatic organisms.

3.1 Fish behavior and movement

In general, fish are more likely to undergo sublethal stress from suspended sediments rather than lethality because of their ability to move away from or out of an area of higher concentration to a lower concentration versus sessile or less mobile species. Therefore, it is important to understand how suspended sediments affect the behavior and physiology over both short- and long-term scales. From our review, three overarching trends appear: preference, physiological adjustment, and avoidance. The consequences of these trends can be observed in a variety of contexts, e.g., social disruption, migratory patterns, displacement of fish, intraspecific aggression, reproductive pairing–spawning success, predator–prey interactions, food web dynamic alternations, larvae disbursement, and settlement (McLeay et al. 1987; Bash et al. 2001; Utne-Palm 2002; Suttle et al. 2004; Muck 2010; Chapman et al. 2014). However, there is a knowledge gap in our overall understanding regarding the relationship between increased sedimentation and behavioral effects (including sensory capabilities, motivation state) on non-salmonids, various migratory species, lotic species, larvae, and fish communities in particular involving the examination of species utilizing various levels of the water column and various life stages for a species that would reside in areas with periodic or chronic sediment loads.

There are relatively few studies that document in situ the ability of fish to avoid suspended sediment plumes and

dredge activity areas and the prevalence of reduced exposure times as a result. Carlson et al. (2001) documented the behavioral responses of salmonids to dredging activities in the Columbia River using hydroacoustics. During dredging operations, out-migrating salmon smolt (*Oncorhynchus* spp., likely fall chinook salmon (*O. tshawytscha*) and coho salmon (*O. kisutch*)) behavioral responses ranged from (1) salmon orienting to the channel margin move inshore when encountering the dredge, (2) most out-migrating salmon passing inshore moved offshore upon encountering the discharge plume, and (3) out-migrating salmon were observed to assume their prior distribution trends within a short time after encountering both the dredging activity and dredge plume (*as cited in* Carlson et al. 2001). In artificial streams, previously unexposed fish juvenile chinook salmon showed a preference of (80 %) for clear water (0 mg/L suspended sediment) in contrast to suspended sediment levels >76 mg/L, and generally avoiding all sediment levels >20 mg/L (Birtwell 1999). In contrast, subadult white sturgeon (*Acipenser transmontanus*) rates of movement, depths, and diel patterns showed little change in response to hopper dredge disposal activities (Parsley et al. 2011). Overall though, volitional fish movement, whether avoidance or displacement and in a few cases preference, will depend upon the "perceived" options available in the water body and an individual's motivation state during elevated suspended sediment loads or dredging activity.

3.2 Foraging and predator–prey interactions

Depending on the foraging strategy of a species, direct exposure to high levels of suspended sediment can disrupt foraging activities or decrease foraging efficacy. The increased turbidity can cause changes in feeding behavior of the fish for the simple reason that the prey may be less visible (Ward 1992). Turbidity, due to the scattering of light, can increase or decrease the contrast between prey and the water column. In the case of some fish larvae, their visual detection of prey increases due to the less inference from light scattering (Utne-Palm 2002). In addition, the protection of larvae from large predators increases from the decreased ability of large visual predators. Thus, in certain cases, turbid environments may offer some benefits for certain species and size groups of fish (planktivores and fish larvae) (Utne-Palm 2002). More commonly though, sedimentation effects on freshwater fish can be graded out by habitat, or life history traits. For example, as sediment and deposited sediments increased, feeding behavior (defined as feeding rate, a reaction distance to food item) decreased in turbidity-tolerant [e.g., northern pike (*Esox lucius*) and largemouth bass (*Micropterus salmoides*)], moderately intolerant [e.g., chinook salmon, rainbow trout (*O. mykiss*)], and intolerant species [e.g., brook trout (*Salvelinus fontinalis*)] (Chapman et al. 2014).

Similar to Chapman et al. (2014), Sullivan and Watzin (2010) found that fish of different life history styles, in this case foraging guilds, have varying tolerances to suspended sediment loads. For example, pumpkinseed (*Lepomis gibossus*, omnivores) showed no significant difference in condition (as measured by Fulton's K Factor) over sediment aggradation (slight, moderate, and severe) or time (14 days) (Sullivan and Watzin 2010). Olsen et al. (1973) reported that rainbow trout feeding activity drops sharply when turbidity surpasses 70 Jackson turbidity units (JTU), or less than 500 ppm by weight for most sediment sources (Noggle 1978). It should be mentioned that the majority of the reviewed literature focuses on species that forage near the surface or within the water column; however, elevated turbidity concentrations can affect the ability of fish to forage on benthic organisms. Sullivan and Watzin (2010) showed that white suckers (*Catostomus commersonii*) and creek chubs (*Semotilus atromaculatus*) experienced a higher mortality compared with pumpkinseed under "severe" sediment aggradation conditions over 28 days. Similarly, Florida pompano (*Trachinotus carolinus*) had reduced foraging success on bean clams (*Donax variabilis*) and mole crabs (*Emerita talpoida*) with increased turbidity (Manning et al. 2013). These results indicate that opportunistic species feeding in several sections of the water column may be more resilient to suspended sediments than more specialized trophic groups (Sullivan and Watzin 2010). These findings indicate that fish foraging success is largely dependent upon their sensory capabilities and adaptive strategies.

Several studies, e.g., Gregory (1993), Gregory and Northcote (1993), Utne-Palm (1999), Bonner and Wilde (2002), Horppila et al. (2004), Rowe et al. (2003), and Shoup and Wahl (2009) provide insightful information regarding the effects of TSS on predator–prey interactions. For example, Gregory (1993), Gregory and Northcote (1993) found that a turbidity threshold of 200 mg/L could reduce dredge-induced salmonid prey–predator reaction changes. Miner and Stein (1996) also reported "changes in predator avoidance," and specifically, reaction distance declined as turbidity increased. Turbidity levels as low as 20 NTU can reduce the overall efficacy of foraging and prey captures in adult and juvenile salmonids (Berg 1982; Bash et al. 2001; Madej et al. 2007), and according to Kemp et al. (2011), other species have had similar effects. However, other studies have indicated that juvenile coho, steelhead, and chinook foraging in slightly to moderately turbid waters (Sigler et al. 1984; Gregory 1988; Bash et al. 2001), and that prey consumption is not significantly affected in species that are adapted to more turbid waters (Kemp et al. 2011).

3.3 Fish physiology and direct physical stress

The deleterious effects of suspended sediments directly on fish physiology are well documented. While increases in sediment load (both suspended and deposited) can have a negative effect across multiple scales of fish communities, from individual level (e.g., spawning success and fry emergence) to the system level (e.g., decreased species richness) (Chapman et al. 2014), the direct cause–effect pathways linking the impacts of sediment loads directly to injury and/or physiological stress is still ambiguous (Nightingale and Simenstad 2001). Rich (2010) provides an informative table of exposure concentrations, durations, and associated mortality.

3.3.1 Sublethal stress

In order to determine suspended sediment effects, Berli et al. (2014) examined metabolic parameters associated with swimming performance in juvenile trout, comparing hatchery strains of rainbow trout (RBT) and a strain of brown trout (*Salmo trutta*; BNT), using three concentrations of calcium carbonate. In general, as turbidity increased, swimming performance decreased, and RBT strains experienced a higher degree of impairment in swimming performance than BNT (Berli et al. 2014). For the groups, indicators of aerobic metabolism (i.e., citrate synthase activities and glucose levels) were elevated, while those of anaerobic metabolism (i.e., plasma lactate and LDH activities) were depressed (Berli et al. 2014). Based on these results, Berli et al. (2014) suggested that acute exposures to environmentally relevant turbidities generated by fine suspended sediment may cause a reduced U_{crit} and that these changes may be related to changes in the utilization of aerobic and anaerobic pathways.

There are many environmental factors that are responsible in determining the magnitude of suspended sediment impact on salmonids including the following: duration and frequency of exposure, water temperature sediment toxicity, fish life stage and life history, particle angularity and size, sediment pulse magnitude and timing, physical condition of biota, and refugia/habitat availability and access (Bash et al. 2001; Muck 2010). Results indicate seasonal changes in the tolerance of salmonids to suspended sediment. For instance, metabolic oxygen demand increases as temperature increases, but water oxygen concentrations decrease (Muck 2010) and may decrease even more during dredging activities where organic material is re-suspended, i.e., associated oxygen requirements during decomposition. Excess suspended sediments can result in significant changes in behavior (Wedemeyer et al. 1984; Schreck et al. 1997; Sutherland 2003), such as feeding (e.g., Berg and Northcote 1985), predator avoidance (Miner and Stein 1996), and modified movement or migration (Carlson et al. 2001); reduced food availability (Kemp et al. 2011); gill trauma (Goldes et al. 1988; Newcombe and MacDonald 1991; Beussink 2007); and increased metabolic costs or energy expenditure shifts (Schreck 2010).

3.3.2 Stressor duration, tolerance, and lethality

The effects of suspended sediments on fish vary across species and depend upon several factors, including the life history and species-specific characteristics (e.g., sediment tolerance), the duration of exposure, frequency of events, the type of sediment (including angularity). Generally, benthic species are more tolerant to suspended sediment than pelagic species (Rogers 1969; Sherk et al. 1974; Noggle 1978). Also, closely related species can express different stress levels at similar exposures, e.g., chum salmon *(O. keta)* fry exposed to suspended sediment concentrations of 28 and 55 g/L resulted in 50 % mortality after 96 h (Smith 1978), whereas the same mortality rate was expressed at lower concentrations 1.2–35 g/L over the same duration for coho, chinook, and steelhead salmon (Noggle 1978). Notably, threshold effects can result in higher mortalities; e.g., rainbow trout in the Powder River (Oregon) died within 3 weeks when the concentration of suspended sediment reached 1000–2500 ppm (Campbell 1954). Table 1 reports suspended sediment mortality effects for several other species. In addition to mortality effects, suspended sediment can impact other aspects of fish behavior and physiology, including growth rate (e.g., whitetail shiner, *Cyprinella galactura*) (Sutherland 2003) or feeding behavior [e.g., rainbow trout (Olsen et al. 1973; Noggle 1978)]. Berry et al. (2003) provide much more detail regarding suspended sediments and effects on fish species.

Understanding the interactions between stressors and a species' life history and physiology can help natural resource managers design management actions that mitigate the effects of the stressor, or assist with the development of therapeutants (Schreck et al. 2001). Given the ebb and flow of disturbance regimes in nature, fish have developed trade-offs for dealing with stress that, in general, affects reproductive fitness by altering gametic or progenic quality (Schreck et al. 2001). For example, disturbance, including handling, can affect the timing of reproduction: in rainbow trout disturbance delays reproduction, whereas tilapia (*Oreochromis niloticus*) either accelerate or completely inhibit reproduction depending on which maturational stage is occurring during the disturbance event (Schreck et al. 2001).

There is currently a lack of holistic-based studies that can identify how the stressors impact fish across scales (Schreck 2010), which makes understanding the total

Table 1 Effects of suspended sediment levels by species

Common Name	Species	Sediment	Concentration	Duration	Mortality (%)	References
Carp	*Cyprinus* spp.	Montmorillonite clay	175,000–225,000 ppm	days	100	Wallen 1951
Cunner	*Tautogolabrus adspersus*	Various sediments	3–300 g/L	12–48 h	50	Noggle 1978
Fourspine stickleback	*Apeltes quadracus*	Various sediments	3–300 g/L	12–48 h	50	Noggle 1978
Golden shiner	*Notemigonus crysoleucas*	Montmorillonite clay	175,000–225,000 ppm	days	100	Wallen 1951
Mummichog	*Fundulus heteroclitus*	Estuary sediment/ fuller's earth	24–169 g/L	24 h	10–90	Noggle 1978
Mummichog	*Fundulus heteroclitus*	Various sediments	3–300 g/L	12–48 h	50	Noggle 1978
Sheepshead minnow	*Cyprinodon variegatus*	Various sediments	3–300 g/L	12–48 h	50	Noggle 1978
Shiner perch	*Cymatogaster aggregata*	Bentonite Clay	0.3–0.9 g/L	10 days	10–50	Peddicord et al. 1975; Noggle 1978
Spot	*Leiostomus xanthurus*	Estuary sediment/ fuller's earth	13–111 g/L	24 h	10–90	Noggle 1978
Striped bass	*Morone saxatilis*	Bentonite Clay	1–5 g/L	10 days	10–50	Peddicord et al. 1975; Noggle 1978
Striped killifish	*Fundulus majalis*	Estuary sediment/ fuller's earth	1–5 g/L	24 h	10–90	Noggle 1978
White perch	*Morone americana*	Estuary sediment/ fuller's earth	3–39 g/L	24 h	10–90	Noggle 1978
Zebrafish	*Danio rerio*	Inorganic limestone	4.8 g/L	4 h	100	Reis 1969

impact of stress on fish populations difficult. Stress response cycles vary not only with the duration and severity of the stress response, but also with the developmental stage of individuals as well as across different physiological, genetic, or reproductive processes. Resisting a stressor and mounting a stress response are energetically costly processes, and the energy required to deal with the stress must be reallocated (generally toward increased oxygen consumption and metabolic rate to deal with the event) (Barton and Schreck 1987; Contreras-Sanchez et al. 1998; McCormick et al. 1998; Muck 2010). If this reallocation reduces energy for reproduction, then there can be population-level consequences. Both the nature of a stressor and its severity can affect fish reproduction in many different ways: for example, evading predators (i.e., an emergency response) or coping with resource limitations due to higher densities are both stressors, but both involve allocating and re-budgeting energy differently, both of which can have impacts at the population level (Schreck 2010); for example, energy is diverted from reproduction to heat-shock protein production (Krebs and Loeschcke 1994; Loeschcke et al. 2013).

Reproduction can be affected by stress in various ways, depending upon when it is experienced in the life cycle and the severity and duration (Schreck 2010). Increased suspended sediment loads can cause physiological, bioenergetic, and behavioral alterations (e.g., delays in spawning) which may in turn affect egg quantity or quality and embryo development (Bash et al. 2001). For example, stress from suspended solids impacts eggs and alevins more than adults (Muck 2010), but other sources of stress can impact ovulation or inhibit reproduction, which can impact both gamete quality and fecundity. Acute stress can have several effects on the reproduction of fish, including reducing egg size and delaying ovulation in females, reducing sperm counts in males, and lowering survival rates for offspring from stressed fish (Campbell et al. 1992), as well as significantly altering relative fecundity, particularly when compared to non-stressed individuals (Contreras-Sánchez et al. 1998). Cumulative stress can also impact reproduction [e.g., chronic confinement stress reduced egg size in rainbow trout and significantly lowered survival rates for progeny from both stressed brown trout and rainbow trout compared to progeny from unstressed controls (Campbell et al. 1994)]. Stress can also impact nutritional quality, which has been correlated with reproductive success [e.g., in wild cod (*Gadus morhua*) lower fecundity resulted from poorer nutritional condition (Lambert and Dutil 2000; Lambert et al. 2000; Schreck 2010)].

Stress induced by suspended sediments can also have impacts at the community level (Waters, 1995), including alterations in habitat (Allan et al. 1997), community diversity and productivity (Dudgeon 2000; Sullivan et al.

2006), and the relative abundance of spawning guilds (Sutherland et al. 2002; Sullivan and Watzin 2010). Further, specialized foragers, such as white suckers, tend to be negatively affected more during sedimentation events, particularly when they are longer in duration, indicating that opportunistic species that feed across the water column may be more resilient to sedimentation than more specialized trophic groups (Sullivan and Watzin 2010). However, the mechanisms responsible for these patterns are complex and still not fully understood (Sullivan and Watzin 2010). In cases where streams are aggraded in patchy distributions, fish can often avoid higher sediment concentrations (Sullivan and Watzin 2010), but if the spatial distribution of suspended sediments is more uniform, then adverse effects via habitat alterations may be spread across multiple life stages (e.g., adults, nest building, egg development, and fry feeding) (Newcombe and Jensen 1996; Galbraith et al. 2006).

Several studies have found that exposure of fish to suspended solids can elicit a primary stress responses (PSR) by increasing both circulating and whole blood cortisol concentrations and levels, respectively (Redding et al. 1987; Humborstad et al. 2006; Sutherland et al. 2008; Rich 2010). There are fewer studies on the secondary stress responses (SSR) in fish exposed to SS and turbidity due to dredging, although three laboratory-based studies were reported in Rich (2010). Types of dredge-related studies conducted where tertiary stress responses (TSR) were used as endpoints consisting of seven field-based and seven laboratory-based studies (Rich 2010). It would seem an important avenue for further research given that the stress hormones, induced by PSR, can affect every organ and function of the body via SSR (Rich 2010). Changes in blood constituents, heart rate, metabolism, and osmoregulation are examples of SSR (Rich 2010). If the body is not able to re-equilibrate from the SSR then TSR result, e.g., lowered resistance to disease, slowed growth rate, and changes in behavior (e.g., avoidance) (Rich 2010).

In most estuaries, average concentrations of total suspended matter can range from a few mg/L to several tens of mg/L, with the higher concentrations occurring near the benthic layer in areas re-suspension (Auld and Schubel 1978). However, during short-term episodic events, like dredging or spoil disposal, concentrations may be greater than several thousand mg/L, particularly in the regions nearest the source of the event. Laboratory studies have indicated that there is a complex set of interactions among species, life stage, concentration, and duration. For example, survival is reduced in larval striped bass and yellow perch (*Perca flavescens*) during 48- to 96-h exposures of 2500 mg/L, but American shad are less tolerant (an exposure of >100 mg/L for the same duration reduced survival); concentrations of over 1000 mg/L affected the hatching success of white perch and striped bass, but lower concentrations had no effect (Auld and Schubel 1978). In another study, when Pacific herring (*Clupea pallasi*) were exposed to concentrations of 250–500 mg/L for the same duration, self-aggregation of the eggs led to both lethal and sublethal impacts (Griffin et al. 2009). However, Kiørboe et al. (1981) reported that no impacts were discovered on Atlantic herring (*C. harengus*) eggs when they were exposed to 5–300 mg/L at different stages of embryonic development.

A recent review of the biological effects of suspended sediments on fish and shellfish was conducted by Wilber and Clarke (2001) (Berry et al. 2003). Berry et al. (2003) synthesized the results of studies that report the dose–response relationships of estuarine aquatic organisms to suspended sediments and then related those findings to sediment conditions associated with dredging projects. Suspended sediment effects on invertebrates include: direct impacts due to abrasion, interference with respiration and ingestion by clogging of filtration mechanisms, and in extreme cases mortality from smothering and burial (Berry et al. 2003). EIFAC (1965) reported harmful levels of solids for Daphnia (*Daphnia* spp.): kaolinite at 102 ppm, montmorillonite at 82 ppm, charcoal at 82 ppm, and pond sediment at 1458 ppm (Bash et al. 2001). The distribution of infaunal and epibenthic species be impacted indirectly through light attenuation affecting feeding efficiency, behavior (avoidance and drift), and habitat alteration occurring from changes in the composition of substrate (Donahue and Irvine 2003; Waters 1995; Zweig and Rabeni 2001; Berry et al. 2003). Increases in suspended sediments (e.g., 120 mg/L) can result in increased drift, significantly altering the distribution of benthic invertebrates in streams (Herbert and Merkens 1961; Berry et al. 2003). Both the duration and degree of exposure (i.e., TSS) are important factors to consider with regard to determining the effects on aquatic organisms (Berry et al. 2003). Waters (1995) considers the effects of increased deposition of sediments on benthic invertebrates as one of the most important concerns within the sediment pollution issue, especially with regard to the dependence of freshwater fisheries on benthic productivity (Berry et al. 2003). Fine sediments, in suspension or when deposited, can negatively impact macrophytes (Yamada and Nakamura 2002; Kemp et al. 2011). Fish are also directly affected by fine sediment either in suspension or deposited on the substrate (Kemp et al. 2011). The suspended or deposited fine sediment can influence physiology and behavior, habitat availability, food supply, and ability to forage efficiently (Kemp et al. 2011). Some salmonid species move towards less turbid water (if available) after short-duration exposures (Berg and Northcote 1985; Kemp et al. 2011).

In freshwater systems, sediment management can have both direct and indirect impacts on species throughout the food web (Kemp et al. 2011). Direct impacts of sediment can include invertebrate mortality (e.g., via smothering) (Kefford et al. 2010). Further, suspended sediments can reduce dissolved oxygen and alter the trophic structure, which can cause the following: a reduction in planktonic and periphytic food sources; increased stress levels which can reduce feeding, growth rates; increased energetic costs; and lower immune system response to viral and bacterial infections (Redding et al. 1987; Shaw and Richardson 2001; Sutherland and Meyer 2007). With respect to setting sediment targets using loadings as a metric, Kemp et al. (2011) compiled five main constraints for identifying meaningful thresholds for freshwater fish, which included dependencies on the catchment (Walling 1995), reaches (Collins and Walling 2007), sediments, taxa, and life stage. Briefly, setting sediment targets requires a holistic, system-level approach. Managing sediments requires consideration of hydrogeomorphic setting of both the watershed and reach, understanding the sediment properties of the target location (e.g., fine clays and silts versus organics; contaminated versus uncontaminated) and the species that are being affected, and what life stages of those species would be affected during the time of the management actions (Berry et al. 2003).

Teasing out the impacts of the interacting components of the above-mentioned constraints can be a significant challenge because not only can exposure impact species and life stages differently, but these impacts can occur at different levels within species/life stages (e.g., genetic, physiological, and reproductive,) and can occur at different exposures. For example, physiological stress [measured as an increase immunoreactive corticosteroid (IRC) levels] was reported for all three life stages of two species [spotfin chub (*Erimonax monachus*) and whitetail shiner] when exposed to 100 mg/L (Sutherland et al. 2008). These results indicate that moderate sediment levels (i.e., 100 mg/L) can impact a species regardless of life stage. Similarly, Ayu (*Plecoglossus altivelis*) had a stress response (measured as changes in cortisol levels) response when it was exposed to concentrations of 200 mg/L for 3 h (Awata et al. 2011). Sediment size, shape, and composition have also been reported to impact freshwater fish (McLeay et al. 1987; Servizi and Gordon 1990; Servizi and Martens 1991; Lake and Hinch 1999; Bray 2000). Extremely angular and round sediments of more than 40 g/L were shown to cause decreased white blood cell concentrations at 96-h exposures in juvenile coho salmon (Lake and Hinch 1999).

Low levels of sediment may result in sublethal and behavioral effects such as increased activity, stress, and emigration rates; loss of or reduction in foraging capability; reduced growth and resistance to disease; physical abrasion; clogging of gills; and interference with orientation in homing and migration (*as cited in* McLeay et al. 1987; Newcombe and MacDonald 1991; Barrett et al. 1992; Lake and Hinch 1999; Bash et al. 2001; Watts et al. 2003; Vondracek et al. 2003; Berry et al. 2003; Muck 2010). Sediment fluxes are critical components of aquatic systems, and their dynamics are multidimensional and complex (Berry et al. 2003; Muck 2010). Maintaining these fluxes requires an understanding of the natural temporal and spatial processes as well as any anthropogenic drivers that can impact the system. For example, seasonal flows resulting from spring snowmelt are natural mechanisms that can mobilize the bed and move silt and sand from the coarse substrate (Osmundson et al. 2002), whereas dam-controlled flows might remove these high flows and affect species dynamics [as in the case of the endangered Colorado pikeminnow (*Ptychocheilus lucius*) which occurs on the heavily damned Colorado river, and whose recruitment has decreased as a result of changes in sediment loads resulting from dam operations] (Osmundson et al. 2002).

The degree of fish population declines is usually associated with the quantity of "fines" within a stream ecosystem (Castro and Reckendorf 1995; Muck 2010). Particles with diameters less than 6.4 mm are generally defined as "fines" (*as cited in* Bjornn et al. 1977; Shepard et al. 1984; Hillman et al. 1987; Chapman 1988; Bjornn and Reiser 1991; Rieman and McIntyre 1993; Castro and Reckendorf 1995b; The Montana Bull Trout Scientific Group (MBTSG) 1998; Muck 2010). There are a variety of negative impacts pertaining to fine sediments in streams, including: (1) loss of habitat for macroinvertebrates, i.e., fish prey (Rabeni et al. 2005; Wood et al. 2005), (2) physiological stress and direct physical damage for both fishes and macroinvertebrates (Newcombe and MacDonald 1991; Sutherland and Meyer 2007), (3) reduction in or elimination of reproductive habitat for benthic and crevice spawning fishes (Burkhead and Jelks 2001; Sutherland 2007), and (4) reductions in the locating and capturing prey ability of fishes due to visual impairment (Barrett et al. 1992; Zamor and Grossman 2007; Hazelton and Grossman 2009). According to Hazelton and Grossman (2009), stream fishes can be harmed by fine sediments through several mechanisms including: (1) decreased prey availability, (2) direct physical harm (Berkman and Rabeni 1987), (3) the risk of increased predation (Miner and Stein 1996), and (4) lowered breeding success (Burkhead and Jelks 2001; Sutherland 2007). Biotic responses to suspended sediment as a stressor are complex because they are dependent on: (1) both direct and indirect ecological effects, (2) species life histories, (3) species traits and differential tolerances, and (4) availability of habitat patch refugia (Schwartz et al. 2011), with refugia area being

affected by suspended sediment itself. For example, combinations of flow and TSS concentration can alter habitat for several species. At average monthly flows and TSS concentrations of 150 mg/L, habitat for bluegill sunfish (*Lepomis macrochirus*) was reduced by half, whereas channel catfish (*Ictalurus punctatus*) and largemouth bass were affected similarly, but at concentrations of 200 mg/L and 100 mg/L, respectively (Stuber et al. 1982; Kundell and Rasmussen 1995).

Species that utilize pools or littoral areas can be impacted at different turbidity levels [i.e., 90 JTU for creek chub, and 180 JTU for green sunfish (*Lepomis cyanellus*)] (Kundell and Rasmussen 1995). Identifying biotic response measures that correlate with sediment stressor gradients is problematic (Nietch et al. 2005; Schwartz et al. 2011), primarily because of the possible multiple stressors that can occur in human impacted watersheds, e.g., temperature, toxic pollutants, hydromodification, nutrient enrichment, habitat alteration, degraded riparian condition, and land cover changes (Wichert and Rapport 1998; Sutherland et al. 2002; Walters et al. 2003; Rashleigh 2004; Halse et al. 2007; Magner and Brooks 2008; Schwartz et al. 2011). It also illustrated that each species' traits and life history patterns have a unique relation to a suspended sediment environmental gradient, a gradient that is frequency and duration dependent (Schwartz et al. 2011). Suspended sediment can harm incubating fish eggs and fry (Cedarholm et al. 1982) and reduce the abundance of insect larvae, a food source for fish, by filling up the larvae's guts or nets with indigestible material (Hynes 1973; Ward 1992).

3.4 Reproduction and epigenetics

Effects of suspended sediment on spawning are found in the scientific literature, although they are mainly associated with the deleterious effects of sediment on gametes and embryo viability. There appears to be limited information in the scientific literature specifically looking at the sublethal effects of suspended sediment and associated environmental conditions, including exposure times and frequencies, regarding fish gametes and transgenerational implications. Sopinka et al. (2014) and Taylor et al. (2015) are some of the few examples of research that has been conducted with intergenerational implications in mind in this case regarding the effects of stress on salmon, albeit not involving suspended sediment. These studies demonstrate an important step toward thinking beyond just the physically observable direct effects on the organisms experiencing the effects and investigate the consequences on subsequent generations as well.

Research by Sopinka et al. (2012) examined the effects of pollution on gametes in wild-caught plainfin midshipman (*Porichthys notatus*), offering an insight into the possible effects on sperm characteristics, egg viability, and embryo survival. In another study, mercury exposure resulted in direct and indirect (transgenerational) effects on mummichog, specifically, reduced male survival, reduced ability of the offspring of exposed fish to reproduce successfully, and altered sex ratios (Matta et al. 1999). Yi et al. (2008) studied the concentrations of heavy metals in fish, invertebrates, sediment, and the water in the Yangtze River during 2006–2007 and found that heavy metals were 100–10,000 times higher in the sediment versus the water (Yi et al. 2008). The concentrations of heavy metals in the tissues of benthic invertebrates were relatively high also (Yi et al. 2008).

Reproductive behaviour and physiology can be negatively impacted by aquatic contaminants (Sopinka 2010). For instance, contaminant mobilization, contaminant leaching, bioaccumulation, and trophic transfer through the food web could occur because of the dredging or disposal of contaminated sediments, but the expression of the impacts thereof in exposed biota may have a lag time (Nightingale and Simenstad 2001). Gamete viability has been used previously as an indicator of reproductive endocrine disruption in fish (Kime and Nash 1999). Sopinka (2010) examined impacts on sperm in plainfin midshipman (*Porichthys notatus*) and round gobies (*Neogobius melanostomus*) and found that living in contaminated areas influenced gamete quality, specifically, greater proportions of dead eggs, greater testicular asymmetry, and shorter sperm heads. Sundberg et al. (2007) found a correlation between hepatic DNA adducts, via polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs), and pollutant burden in Northern pike and European perch (*Perca fluviatilis*) eggs revealing a threat to early life stages of fish. In Texas, the Houston Ship Channel (HSC) and upper Galveston Bay (GB) have sediments contaminated with dioxin (Yeager et al. 2010), and elevated dioxin concentrations have been detected in fish and crabs, i.e., Houston Ship Channel (HSC) in 1990 (Crocker and Young 1990; Yeager et al. 2010). Recent research has shown that the most significant dioxin reservoir is located in the bottom sediments (Suarez et al. 2005; Yeager et al. 2007, 2010). Sediment re-suspension, such as that associated with dredging activities, may re-introduce dioxins into the food chain (Yeager et al. 2010).

The impacts of dredging operations on aquatic organisms are still poorly understood. Some studies have shown that dredging contaminated sediments increases particulate-matter-associated contaminants in waters next to or near to the dredge, producing deleterious effects on species that occupy those areas. (e.g., Bellas et al. 2007; Bocchetti et al. 2008; Engwall et al. 1998; Sundberg et al. 2007; Sturve et al. 2005; Yeager et al. 2010). In order to prevent

or minimize exposure of certain organisms to the effects of dredging activities, environmental windows (EWs) have been put in place in certain circumstances by regulatory and resource agencies (Suedel et al. 2012). EWs can be described as certain times when dredging and dredge material placement activities can be performed (NRC, 2001; Suedel et al. 2012). In contrast, other studies, e.g., Suedel et al. (2012, 2014) reported that there were no statistically significant effects on walleye (*Sander vitreus*) egg viability and hatching rates nor detrimental effects to fry exposed to SS exposures mimicking sediment re-suspension during dredging operations. However, the authors of the aforementioned study do mention that a slight reduction in egg viability occurred at 500 mg/L TSS and that more research is needed regarding the potential sublethal effects in general. Further, Arambourou et al. (2014) noted that morphological abnormalities could appear after several generations of exposure. It is now known that some toxics, such as endocrine disruptors, can contribute to transgenerational developmental effects in aquatic organisms, such as in the Japanese rice fish, i.e., medaka (*Oryzias latipes*) (Gray et al. 1999; Zhang et al. 2008), leading to an increase in morphological (phenotypic) abnormalities in the offspring derived from the exposed parents (Arambourou et al. 2014).

It is known that sediments can serve as a carrier of many metals and toxic compounds, e.g., Pb, Cd, Zn, Cu, Al, Fe, Mn, Cr, and Ni (see e.g., Novotny and Chesters 1989; Kundell and Rasmussen 1995). Rivers contaminated by metals and organic substances have often been reported to be associated with an increased incidence of phenotypic defects, such as phenodeviation and fluctuating asymmetry (FA), particularly in invertebrates (Al-Shami et al. 2011; Bonada and Williams 2002; Groenendijk et al. 1998; Arambourou et al. 2014). Deposition of organic sediments can result in anaerobic conditions in rivers and streams by increasing the sediment oxygen demand (SOD) (Kundell and Rasmussen 1995). DNA damage in the male germ line has been linked to poor semen quality, low fertilization rates, impaired pre-implantation development, increased abortion, and an elevated incidence of disease in the offspring, including childhood cancer (*as cited in* Lewis and Aitken 2005). In addition, the cellular machinery that allows these cells to undergo complete apoptosis is progressively lost during spermatogenesis, so that the advanced stages of germ cell differentiation cannot be deleted, even though they may have proceeded some way down the apoptotic pathway (Gorczyca et al. 1993; McVicar et al. 2004). Consequently, the ejaculated gamete may exhibit genetic damage to both its nuclear and mitochondrial genomes (Sawyer et al. 2001, 2003). For instance, a study using zebrafish by Gosálvez et al. (2014) found that although sperm DNA fragmentation did not significantly influence oocyte fertilization capacity, it significantly and negatively affected later embryo development and overall reproductive success.

Many studies, e.g., Franklin and Mansuy (2010), Gillette et al. (2014), Christopher (2014), Gapp et al. (2014), and Missios et al. (2014), in mammals have shown that several stress-induced outcomes, such as DNA methylation and telomere length, can affect an individual's health, thereby impacting successive generations and population dynamics. Exposure to environmental chemicals and heavy metals such as BPA and Cd can negatively affect both male and female reproduction, alter behavior, and act as a carcinogen with short-term and long-term effects that typically occur through epigenetic mechanisms such as DNA methylation or noncoding RNAs, as has been observed in several species (e.g., Dhimolea et al. 2014; Mileva et al. 2014; Liu et al. 2014; Tellez-Plaza et al. 2014; Ray et al. 2014). Chemicals and heavy metals can cause epigenetic changes (e.g., DNA methylation) and genetic changes (e.g., telomere length) resulting in decreased gamete production and gamete quality, thereby negatively influencing population dynamics (Franklin and Mansuy 2010; Gillette et al. 2014). Also, these epigenetic marks and subsequent adverse effects can be transferred to the offspring through the gametes (Weigmann 2014; Wei et al. 2015; Gapp et al. 2014).

3.5 Models

Over the last decade, quantitative modeling has been used increasingly to determine the impact of dredging operations on fish (Clarke and Wilber 2000; Rich 2010), although there have been few focused modeling studies that examine the system-level impacts of dredging on fish populations. Existing models have been highly criticized as unreliable (Gregory et al. 1993; Clarke and Wilber 2000; Rich 2010), based on high levels of uncertainty associated with data used for model parameterizations, a lack of understanding of threshold values for minimum and maximum durations/concentrations below or above which impacts would not occur, subjectivity of expert opinion and index values used in model parameterizations, unrealistic sediment concentrations used in model scenarios, not including other environmental parameters such as water temperature, dissolved oxygen, particle size, and particle shape, and lack of field-based empirical studies on which to validate the models, without which it is difficult to derive the driver–stressor–response pathways (Newcombe and Jensen 1996; Burkhead and Jelks 2001; Rich 2010).

In order to quantitatively examine the biological and ecological impacts of dredging-created sedimentation, a model must provide the structure to quantify the extent and timing of mixing along the sediment–water interface. However, developing this type of model is complicated

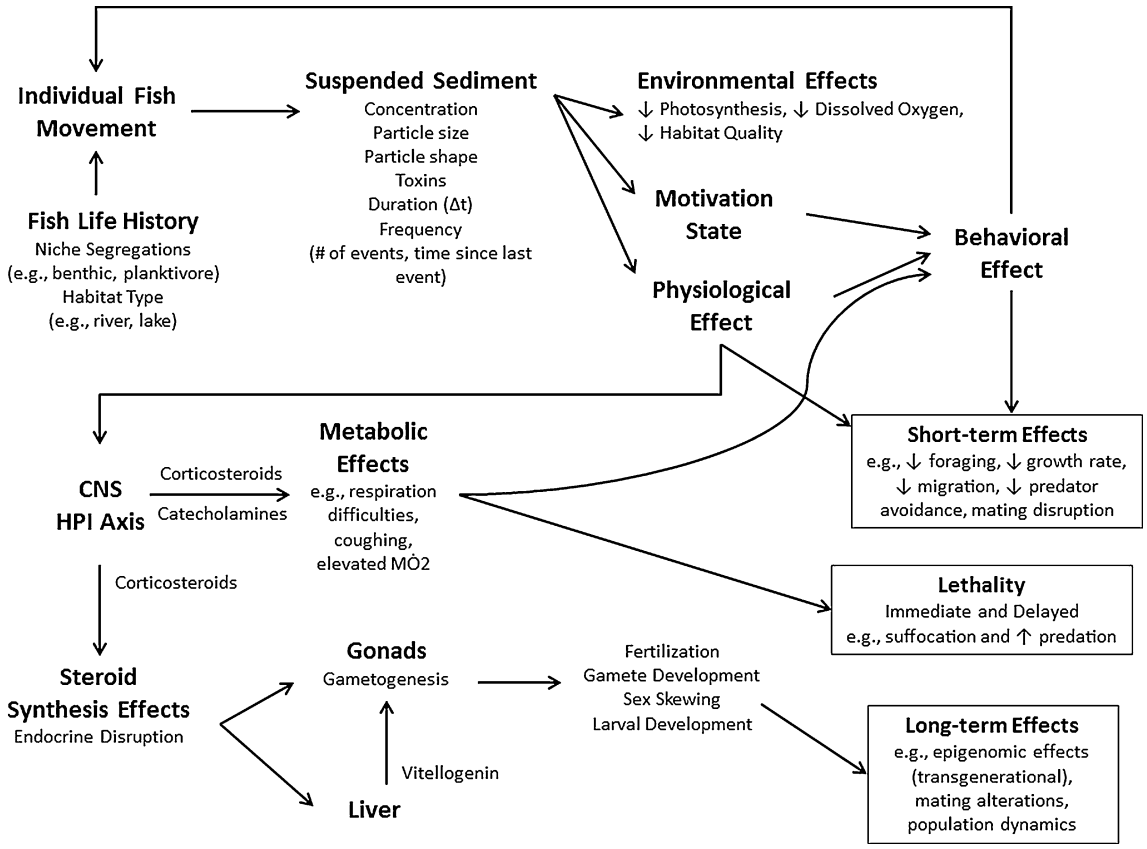

Fig. 1 Conceptual model of the effects of suspended sediments on fish

because it is difficult to resolve the issues on determining available sediment concentrations in the water column and how those concentrations will affect fish dynamics. In the Great Lakes, walleye eggs and larvae have been reported to be affected by sedimentation, but empirical datasets are largely lacking and extensive laboratory dose–response data are unavailable (Germano and Cary 2005), aside from a few more recent studies [e.g., Suedel et al. (2012, 2014)]. For salmonids, Lisle and Lewis (1992) developed a model focusing on survival of salmonid embryos, and they concluded that further research was needed to resolve the interactions among sediment transport, the inter-gravel environment, and embryo survival. Germano and Cary (2005) indicated that the fidelity and scale of available sedimentation data do not have fine enough resolution to predict impacts on the early stages of fish development. However, the effects of embeddedness have been described on hatching salmonids (Waters 1995) and this has helped with guidance to maximize salmonid production based on bed composition (Lotspeich and Everest 1981; Caux et al. 1997; Germano and Cary 2005). Further, there have been numerous studies that summarized both direct and indirect effects of sediments on other species (as summarized in Berry et al. 2003), but few have been synthesized into large-scale models. Dynamic energy budget (DEB) models (Noonburg et al. 1998; Nisbet et al. 2000) are designed to predict effects of stress on organism growth and survival, but have yet to be applied to dredging-created impacts on fish populations (Germano and Cary 2005).

Newcombe and McDonald (1991) proposed a dose–response model, but the major problem with the proposed model was the simplicity, subjectivity of ranked responses, and lack of a well-defined mathematical model (Bray 2000). Gregory et al. (1993) had several criticisms of the Newcombe and McDonald (1991) model. First, there was a large variance in the data compiled by Newcombe and McDonald (1991), inherently reducing the model's predictive power of the model. A validation procedure was also lacking in the Newcombe and McDonald (1991) model for comparing the actual field observations with model predictions (Bray 2000). Another criticism of the Newcombe and McDonald (1991) model was that it did not have established threshold durations or concentrations beyond which impacts would not occur (Bray 2000). Gregory et al. (1993) pointed out that suspended sediment impacts will be variable not only with species, but also with life stage (Bray 2000), an aspect lacking in the Newcombe and McDonald (1991) model. Further,

significant variables like water temperature and sediment size were not included in Newcombe and McDonald's (1991) model formulation, although such variables may play a major role in an organism's response to suspended sediments (Gregory et al. 1993).

In a subsequent modeling effort, Newcombe and McDonald (1993) reformulated the model with regard to the listing of ranked severity of ill effects, defining specific thresholds levels, behavioral effects, sublethal effects, and lethal effects (Bray 2000). Newcombe and McDonald (1993) also utilized data from a study by Servizi and Martens (1992) to show that impact on biological response for a particular species can vary by life stage. By pooling data from the scientific literature, from approximately 264 field studies, Newcombe and Jensen (1996) improved the SEV model in their last revision (see lethal and paralethal effects, concentrations and duration data for different species). Bray (2000) points out that definitions of appropriate recovery times associated with exposure of salmon species to SSC levels, i.e., thresholds, would be very useful for assessing biological impact from the Newcombe and Jensen (1996) model results, since this aspect was not accounted for in the model.

A clearer understanding of dredging, and consequently suspended sediments, effects to fish requires a synthesis of life history strategies, behavior and movement, physiology, organismal-level short-term effects, and ecological endpoints (i.e., lethality, epigenetic effects). Figure 1 illustrates the conceptual model of the effects of suspended sediments on fish that was developed based on the available literature.

4 Conclusion

Future modeling efforts are required to build upon the criticisms of model shortcomings and simplistic assumptions utilized in some of the past modeling efforts. Given recent advances in spatially explicit agent-based modeling, pattern-oriented modeling, and inverse modeling techniques, more realistic and informative models can be constructed and implemented that can project long-term effects (positive or negative) of sediment interactions with aquatic species. However, explicit regard to not only the organism itself but also transgenerational implications should be investigated and included in future modeling efforts.

Future efforts should focus on developing an understanding of whether environmental disturbances like dredging may lead to epigenetic changes, which can lead to population effects. Such research will be useful in decreasing uncertainty surrounding potential long-term effects of chemicals and heavy metals in the environment in both risk assessments and life cycle analyses. TSS and stress response information with regard to transgenerational and epigenetic implications is needed for more informed long-term management, as well as for assessing if cryptic, physiological non-observable manifestations of stress could threaten fish populations as well as species.

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Compliance with Ethical Standards

Conflict of interest The authors declare that they have no conflicts of interest.

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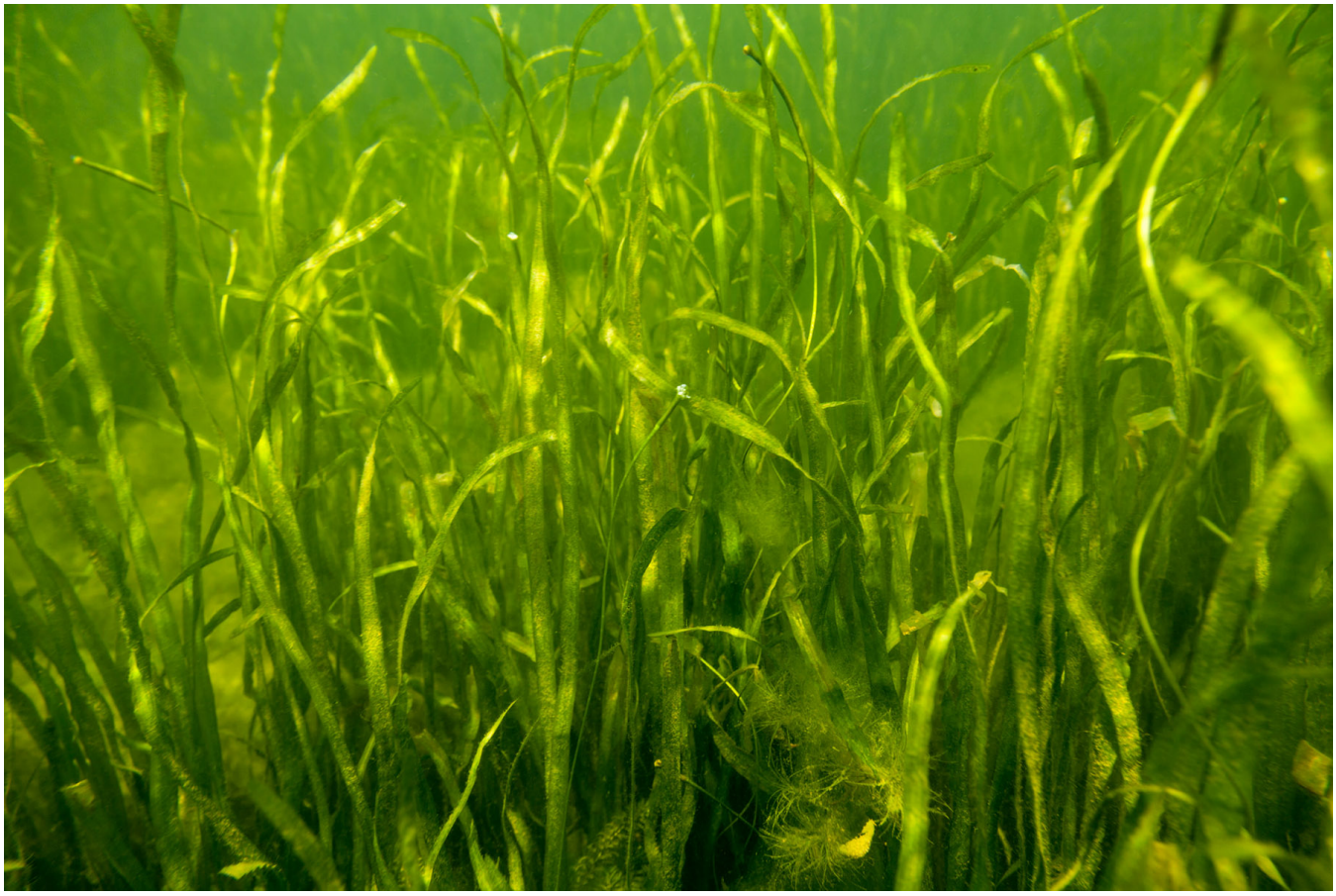
Exhibit D

Source: https://www.chesapeakebay.net/issues/bay_grasses

Underwater Grasses

Underwater grasses grow in the shallow waters of the Bay and its streams. They provide food and habitat to wildlife, add oxygen to the water and trap sediment and nutrient pollution.



Overview

Underwater grasses—also known as submerged aquatic vegetation or SAV—are plants that grow in the shallow waters of the Chesapeake Bay and its streams, creeks and rivers. Underwater grasses are a critical part of the Bay ecosystem: they provide wildlife with food and habitat, add oxygen to the water, absorb nutrient (/issues/nutrients) pollution, trap sediment

(/issues/sediment) and reduce erosion. Like all plants, underwater grasses need sunlight to grow, which makes improving water clarity (/discover/ecosystem/water_clarity) an important step in underwater grass restoration.

What are underwater grasses?

Also known as submerged aquatic vegetation, or SAV, underwater grasses are plants that grow in the shallow waters of the Chesapeake Bay and its streams, creeks and rivers, and are a critical part of the Bay ecosystem.

Why are underwater grasses important?

Underwater grass beds are critical to the Chesapeake Bay ecosystem. They offer food to small invertebrates and migratory waterfowl; shelter young fish and blue crabs; and keep our waters clear and healthy by absorbing excess nutrients, trapping suspended sediment and slowing shoreline erosion.

These grasses also act as an excellent measure of Chesapeake Bay health. Although underwater grasses are sensitive to pollution, they respond fairly quickly to improvements in water quality. This means their abundance is a good indicator of restoration progress. You can watch changes in underwater grass abundance over time using this interactive map (http://www.chesapeakebay.net/visualization/baygrasses/).

Providing food, habitat and oxygen

Bay grass beds provide food and shelter to a number of wildlife species, many of which depend on each other for survival.

- Microscopic zooplankton feed on decaying underwater grasses, ridding grass beds of waste. In turn, these zooplankton become food for larger critters.
- Small invertebrates (/discover/field-guide/all/invertebrates/all)–including barnacles (/discover/field-guide/entry/barnacles), sponges, sea slugs (/discover/field-guide/entry/sea_slugs) and sea squirts (/discover/field-guide/entry/sea_squirt)–feed on and attach themselves to the stems and leaves of underwater grasses. Small crustaceans consume harmful algae that might otherwise grow on underwater grasses and stunt their growth.
- Young crabs and fish–including spot (/discover/field-guide/entry/spot), croakers (/discover/field-guide/entry/atlantic_croaker), weakfish (/discover/field-guide/entry/weakfish), Atlantic menhaden (/discover/field-guide/entry/atlantic_menhaden), white perch (/discover/field-

guide/entry/white_perch) and American shad (/discover/field-guide/entry/american_shad)–find protective nurseries in underwater grass beds. Scientists have found 30 times more juvenile blue crabs in underwater grass beds than in areas with no grasses.

- Small fish dart between underwater grasses as they hunt for prey and hide from predators.
- Molting blue crabs (/discover/field-guide/entry/blue_crab) seek refuge in underwater grass beds while their soft shells make them vulnerable to predation.
- Migratory waterfowl feed on underwater grasses and the animals that live in underwater grass beds.

Underwater grasses also add oxygen to the water during photosynthesis. Underwater critters need oxygen to survive.

Keeping the water clear and healthy

Underwater grass beds help keep the Chesapeake Bay clean and healthy by:

- Absorbing excess nutrients like nitrogen and phosphorous;
- Trapping particles of sand, silt and sediment, which might otherwise cloud the water and suffocate shellfish; and
- Reducing erosion by slowing water currents, anchoring bottom sediment in place and softening waves that break along the shoreline.

How do underwater grasses grow?

Just like plants on land, underwater grasses go through photosynthesis to convert sunlight into food. Sunlight, therefore, is the most important factor determining grass survival. Water must be clear enough for sunlight to pass through it and reach the grasses that grow on the bottom of rivers, streams and the Chesapeake Bay.

How does pollution affect underwater grasses?

Underwater grass growth (/state/underwater_grasses) is hindered by pollutants that cloud the water. These pollutants include excess nutrients (issues/nutrients)—which fuel the growth of dense algae blooms—and suspended particles of sand, silt and sediment (/issues/sediment).

Healthy underwater grass beds can trap and absorb some of this nutrient and sediment pollution, but too much of it can block sunlight from reaching the plants.

How does weather affect underwater grasses?

Extreme weather (/issues/climate_change), including high temperatures or excess rainfall, can harm underwater grasses.

Temperature

Some species, like eelgrass (/discover/field-guide/entry/eelgrass), cannot grow in water that is too warm. In 2005, high temperatures caused large beds of eelgrass in the lower Chesapeake Bay to die. It can take several years for underwater grass beds to recover from these kinds of large-scale losses.

Precipitation

Precipitation—and the water-clouding pollution it pushes into rivers and streams—has a big influence on underwater grass growth.

- Higher than average rainfall can push nutrient and sediment pollution into the Bay and its rivers and streams. In 2012, scientists recorded a 21 percent decline (/news/blog/chesapeake_bays_underwater_grasses_decline_in_2012) in underwater grass abundance, attributed in part to the strong storms—like Hurricane Irene and Tropical Storm Lee—seen in the late summer and fall of 2011.
- Lower than average rainfall can result in clearer water, which can boost underwater grass growth.

How are underwater grasses being restored?

Modern science allows researchers to target underwater grass restoration (/state/underwater_grasses) to those areas where grasses once grew. But poor water quality, irregular weather and a lack of funding have slowed restoration progress.

Chesapeake Bay Program partners use four initiatives to restore and maintain the health of underwater grasses: improving water clarity (/discover/ecosystem/water_clarity), planting underwater grasses, protecting existing grass beds and enhancing underwater grass-related education and outreach.

Improving water clarity

Improved water clarity is critical to underwater grass restoration because clouded water can block sunlight from reaching aquatic plants. When water is clear, more sunlight can reach the bottom of rivers, streams and the Chesapeake Bay, fueling the growth of new grasses and the expansion of existing grass beds.

Maryland, Virginia and the District of Columbia have outlined water clarity standards for the shallow waters where underwater grasses grow or once could be found. These standards list the amount of sunlight that must be able to pass through the water.

Pollution reduction efforts—including the Bay "pollution diet (/what/programs/total_maximum_daily_load)," or Total Maximum Daily Load (TMDL)—aim to slow the flow of nutrients and sediment into the Bay and its waterways. Upgrades to wastewater (/issues/wastewater) treatment plants and on-farm conservation practices (/issues/agriculture), for example, are expected to improve water clarity.

Planting underwater grasses

Although underwater grasses can naturally colonize an area faster than we can spread them through planting, planting underwater grasses can lead to restoration success, enhancing the natural expansion of healthy grass beds.

The Maryland Department of Natural Resources (http://dnr.maryland.gov/) (DNR), for instance, led two large-scale eelgrass restoration projects on the Patuxent and Potomac rivers between 2003 and 2007. Scientists found that distributing seeds collected from healthy grass beds elsewhere could accelerate the natural expansion of weaker eelgrass beds. Scientists need sustained funding and adequate plant and seed supplies to continue this kind of work.

Protecting existing grass beds

Underwater grass beds can be damaged by a number of things, including human activities, invasive species and climate change.

- Shellfish dredges and boat propellers can pull underwater grasses up from the bottom of rivers, streams and the Bay. In shallow waters, "scars" from these human activities are often visible across grass beds.
- Invasive species (issues/invasive_species) can threaten existing underwater grass beds. Mute swans (/discover/field-guide/entry/mute_swan), for instance, can eat more than eight pounds of grasses in a single day, pulling plants up from their roots and depleting entire grass beds. And water chestnut (/discover/field-guide/entry/water_chestnut) floats on the water's surface, blocking sunlight from reaching grasses growing underneath.
- Rising temperatures that accompany climate change (issues/climate_change) could make the Bay an unsuitable habitat for certain underwater grass species. And the flooding and shoreline erosion that accompany sea level rise could lead to a further decline in water clarity.

Enhancing education and outreach

Grasses for the Masses (http://www.cbf.org/how-we-save-the-bay/programs-initiatives/virginia/grasses-for-the-masses), started by the Chesapeake Bay Foundation (http://www.cbf.org/) (CBF), allows Virginia residents to raise wild celery in their homes. The grasses are then planted at restoration sites in Virginia rivers that flow into the Bay.

Current restoration goals

As part of the *Chesapeake Bay Watershed Agreement* (/what/what_guides_us/watershed_agreement), Chesapeake Bay Program partners have committed to the goal (https://www.chesapeakebay.net/what/goals/vital_habitats) of achieving and sustaining 185,000 acres of underwater grasses in the Bay, with a target of 90,000 acres by 2017 and 130,000 acres by 2025. In 2018, there were an estimated 91,559 acres (https://www.chesapeakebay.net/news/blog/despite_record_rainfall_underwater_grass_abundance_remains_strong) of underwater grasses in the Bay. Learn more about their current restoration levels on ChesapeakeProgress (https://www.chesapeakeprogress.com/abundant-life/sav).

Take Action

For Chesapeake Bay restoration to be a success, we all must do our part. Our everyday actions can have a big impact on the Bay. By making simple changes in our lives, each one of us can take part (/action/howtotips) in restoring the Bay and its rivers for future generations to enjoy.

To support underwater grasses in the Bay watershed, boaters can follow posted speed limits and no-wake laws to avoid harming grass beds. Boaters can also steer clear of any grasses growing in shallow waters.

Related Articles by Category

(https://www.chesapeakebay.net/news/blog/bay_health_impacted_by_record_flows)

Bay health impacted by record flows

(https://www.chesapeakebay.net/news/blog/volunteer_as_a_chesapeake_bay_sav_watcher)

Volunteer as a Chesapeake Bay SAV Watcher

(https://www.chesapeakebay.net/news/blog/despite_record_rainfall_underwater_grass_abundance_remains_strong)

Despite record rainfall, underwater grass abundance remains strong

Exhibit E

Water Science School

Biological Oxygen Demand (BOD) and Water

You don't often think that water bodies contain oxygen, but water does contain a small amount of dissolved oxygen. A small amount, but it is essential for life in the water. Biological oxygen demand (BOD) generally represents how much oxygen is needed to break down organic matter in water.

• **Water Science School HOME** • **Water Properties topics** • **Water Quality topics** •

Biological Oxygen Demand (BOD) and Water

Biochemical oxygen demand (BOD) represents the amount of oxygen consumed by bacteria and other microorganisms while they decompose organic matter under aerobic (oxygen is present) conditions at a specified temperature.

When you look at water in a lake the one thing you don't see is oxygen. In a way, we think that water is the opposite of air, but the common lake or stream does contain small amounts of oxygen, in the form of **dissolved oxygen**. Although the amount of dissolved oxygen is small, up to about ten molecules of oxygen per million of water, it is a crucial component of natural water bodies; the presence of a sufficient concentration of dissolved oxygen is critical to maintaining the aquatic life and aesthetic quality of streams and lakes.

The presence of a sufficient concentration of dissolved oxygen is critical to maintaining the aquatic life and aesthetic quality of streams and lakes. Determining how organic matter affects the concentration of dissolved oxygen (DO) in a stream or lake is integral to water- quality management.

Status - Completed

Contacts

Ask USGS

https://answers.usgs.g

Explore More Science:

Biological oxygen demand
eutrophication
Dissolved oxygen
BOD
Water Quality
Water Properties
Chemical Water Properties
Surface-Water Quality Issues
Water-Quality Properties
Water

The decay of organic matter in water is measured as biochemical or chemical oxygen demand. Oxygen demand is a measure of the amount of oxidizable substances in a water sample that can lower DO concentrations.

Certain environmental stresses (hot summer temperatures) and other human-induced factors (introduction of excess **fertilizers** to a water body) can lessen the amount of dissolved oxygen in a water body, resulting in stresses on the local aquatic life. One water analysis that is utilized in order to better understand the effect of bacteria and other microorganisms on the amount of oxygen they consume as they decompose organic matter under aerobic (oxygen is present) is the measure of biochemical oxygen demand (BOD).

Determining how organic matter affects the concentration of dissolved oxygen in a stream or lake is integral to water-quality management. BOD is a measure of the amount of oxygen required to remove waste organic matter from water in the process of decomposition by aerobic bacteria (those bacteria that live only in an environment containing oxygen). The waste organic matter is stabilized or made unobjectionable through its decomposition by living bacterial organisms which need oxygen to do their work. BOD is used, often in **wastewater-treatment plants**, as an index of the degree of organic pollution in water.

Learn more about biological oxygen demand and other related water topics:

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Water Properties Information by Topic

Looking at water, you might think that it's the most simple thing around. Pure water is practically colorless, odorless, and tasteless. But it's not at all simple and plain and it is vital for all life on Earth. Where there is water there is life, and where water is scarce, life has to struggle or just "throw in the towel." Continue on to learn about dozens of water properties.

Contacts: Ask USGS

SUMMARIES OF WATER POLLUTION REPORTING CATEGORIES

This document includes summaries of 34 general reporting categories used for EPA ATTAINS data on polluted waters. The summaries were developed for non-technical audiences to explain clearly what the category is, where the pollution comes from, how it can harm the environment or human health, and what individuals can do to help reduce the problem. These summaries of ATTAINS reporting categories also appear along with simplified common category names in ***How's My Waterway***, a local-scale search application that retrieves ATTAINS data and translates it for general audiences. Simplified names from ***How's My Waterway*** appear in parentheses after the ATTAINS name in the coming pages.

ALGAL GROWTH (EXCESS ALGAE) *can occur when too many nutrients, warm water temperatures, and reduced flow trigger the overgrowth of naturally occurring algae into thick mats on or in the water. Blooms of algae can harm aquatic life by clogging fish gills, reducing oxygen levels, and smothering stream and lake beds and submerged vegetation. Some algae blooms can produce poisons that harm human health, pets, wildlife, and livestock when swallowed.*

What you can do: People can help reduce algae blooms in their local waters by using lawn and plant fertilizer sparingly and never before storms, regularly checking and pumping septic tanks, never dumping plant or animal waste in a waterway, disposing of pet waste in the trash, pumping boat waste to an onshore facility, and planting native plants to reduce excess nutrients entering waterways. Learn more about harmful freshwater algae EXIT Disclaimer and marine algae, and how to reduce nitrogen and phosphorus pollution that causes excess algae growth.

Summary: Ranging from microbes to large seaweeds, algae are a natural part of the plant life in fresh and salt waters. They can become a problem when high nutrients and light, warmer temperatures, and low water flow result in very rapid growth. Runoff from over-fertilized lawns and croplands, leaking septic systems, wastes from animal feedlots, pets, industry, untreated sewage overflow, removal of shoreline plants, and reduced water flow due to irrigation or drinking water withdrawal all can contribute to a bloom. Algae blooms can harm aquatic life by clogging the gills of fish and small aquatic animals, reducing oxygen in the water, or by smothering corals and submerged aquatic vegetation. Algae blooms can also discolor the water, form huge, smelly piles on beaches, or cause drinking water, fish, and shellfish to taste bad. A small percentage of algae produce poisons that can cause illness in humans, pets, fish, livestock, and birds, which could result in death. Economic concerns associated with harmful algae blooms include increased drinking water treatment costs, loss of recreational and tourism income, loss of shellfish and fisheries jobs and food products, and livestock sickness or deaths. Coastal harmful algae blooms have been estimated to result in economic impacts to the United States of at least $82 million each year. Due to the potential human health risks, freshwater algae toxins are on the EPA drinking-water contaminants list, and fish and shellfish advisories are frequently posted in coastal areas. Around 900 waters have been reported in this pollution category nationwide, and several thousand more waters reported as polluted by nitrogen and phosphorus (nutrient) pollution or low dissolved oxygen can also involve algal growth problems.

AMMONIA *occurs naturally in water in trace amounts, but too much ammonia from fertilizers, sewage and other wastes can be poisonous to fish, especially when water temperature and pH are high. Ammonia can also cause heavy plant growth, foul odors, and low oxygen levels that can interfere with use for fishing, swimming and water supplies.*

What you can do: People can help reduce ammonia/nitrogen pollution by applying the correct amount of fertilizer on lawns and not applying it before storms, never dumping manure in or near a stream, picking up and disposing of pet waste in the trash, regularly pumping out septic tanks, and pumping boat waste to an onshore facility. Read more about ammonia pollution effects and what you can do to help reduce ammonia pollution.

Summary: Ammonia occurs naturally and is used in small amounts by plants for growth, but too much of it becomes poisonous to aquatic life especially in higher water temperatures and pH (water that is more basic than acidic). Ammonia is a common cause of fish kills and can harm people's health after it is

converted to nitrate by bacteria in the water. High nitrates in groundwater used for drinking have been linked to potentially fatal oxygen levels in babies, known as "blue-baby syndrome." Also, excess ammonia can cause heavy growth of harmful algae, which can cause illness in humans if swallowed during recreational activities such as swimming. Too much ammonia can also cause oxygen-poor waters, since dissolved oxygen in water is used up by bacteria and other microbes in converting ammonia into their food. Common man-made sources of ammonia pollution include fertilizer production and use, manure application to farmland, septic seepage, concentrated animal feeding operations, untreated sewage overflow, and animal and industrial waste. Around 400 waters have been reported as polluted by ammonia. However, ammonia pollution also plays a big role in nitrogen and phosphorus pollution, which is currently the third highest reported cause of water pollution in the US affecting over 6,000 waterways.

BIOTOXINS (BIOLOGICAL POISONS) *are toxins produced by aquatic plants, animals, and microbes that can sicken or even kill fish, shellfish, pets, livestock, wildlife, and people when swallowed or contacted. The leading producers of these poisons are blue-green algae, which can bloom into thick mats when high temperatures, still water, low water levels, and high nutrient levels are found.*

What you can do: People can help reduce the occurrence of toxic algae in their local waters by using lawn and plant fertilizer sparingly and never before storms, regularly checking and pumping out septic tanks, never dumping plant or animal waste in a waterway, disposing of pet waste in the trash, pumping boat waste to an onshore facility, and planting native plants near shores to reduce nutrient runoff into waterways. Learn more about harmful algal blooms EXIT Disclaimer, their toxins EXIT Disclaimer, and ways to reduce nitrogen and phosphorus pollution that causes excess algae.

Summary: Biological poisons (biotoxins) are water pollutants produced by microbes, animals or plants that can cause illness or death in humans, pets, fish, livestock, and birds. Most of the 80 waters reported in this category nationwide contain toxins produced by blue-green algae. Several thousand more waters are affected by nitrogen and phosphorus (nutrient) pollution, algae growth, or low dissolved oxygen, which can be associated with a potential biotoxin problem. Blue-green algae occur naturally in smaller numbers, but can become a problem when high nutrients and light, warmer temperatures, and/or low water flow, resulting in very rapid growth that creates dense blue-green algae blooms. Runoff of fertilizers on lawns and croplands, leaking septic systems, wastes from concentrated animal feeding operations, livestock farming, pets, and industry, untreated sewage overflow, removal of shoreline plants, and altered water flow for irrigation, municipal water supplies and industry all can contribute to cause a harmful bloom. Exposure to toxins from blue-green algae may occur through swallowing tainted water or fish, inhaling water vapor near a bloom, or contacting polluted water during recreational activities such as swimming. Economic concerns associated with harmful algae blooms include increased drinking-water treatment costs, loss of recreational and tourism revenue, loss of shellfish and fisheries revenue, and livestock sickness or death. Pets and wildlife have died after drinking from waterways with blue-green algae blooms. Due to the potential human-health risks, freshwater algae toxins are on the EPA drinking-water contaminants list, and fish and shellfish advisories are frequently posted in coastal areas with toxic algae problems.

CAUSE UNKNOWN *is a reporting category used when a state has detected degraded conditions in a waterway but has reported no specific details about those conditions or the pollution that caused them.*

What you can do: Your state water program may have more recent information on pollution cause, or added information not reported to EPA about your waterway. Contact your state water program to ask, or to report anything about possible causes that you may have observed. See EPA's CADDIS website for information on scientific methods for solving unknown causes.

Summary: This reason for reporting a degraded waterway means that a state has monitored and detected degraded conditions in a waterway, but has reported no specific details about those conditions or the pollution that caused them. About 1,300 waters are in this category as of the most recent state reporting cycles. Waters can be moved to other pollution categories as more is learned about the actual causes. The degraded conditions observed by the state but not reported may have included degraded fish or invertebrate communities, degraded aquatic habitat, or possibly other effects. Due to the uncertainty about conditions, causes, and sources, it is difficult to generalize about this category's potential effects on human health and beneficial uses or environmental impacts, or provide links for additional detailed information.

CAUSE UNKNOWN - FISH KILLS *-- large numbers of dead fish in a localized area – may be due to water conditions such as low flow, high temperatures or low oxygen levels, or to fish diseases or spills of oil or toxic substances.*

What you can do: People can help by never dumping anything for any reason in a stream or lake, and reporting evidence of fish kills immediately to a state water quality or fisheries management office.

Summary: When unusual numbers of dead fish are found in one place or along a water body, the incident is referred to as a fish kill. Usually fish kills are due to low oxygen or a contaminant in the water, not enough water, or a disease. Most waters with fish kills due to a known pollutant or other cause are reported under the pollutant type. The cause of death is sometimes unknown or unreported. This category includes 83 waters reported for fish kills of unknown cause. Fish kills may be due to an isolated event such as a toxic spill into the water, but also can happen repeatedly under recurring conditions such as low flow or depleted oxygen. Fish kills may not affect human health, but they often mean reduced or lost fishing opportunities for up to several years. Rotting fish also degrades several other waterside recreational uses. These losses of beneficial use can hurt local economies that involve recreation. A fish kill also harms the environment by reducing or removing a major part of the water body's food chain, and this may sometimes enable less desirable aquatic life to dominate.

CAUSE UNKNOWN - IMPAIRED BIOTA (DEGRADED AQUATIC LIFE) *means that the community of aquatic animals (fish, reptiles, amphibians, aquatic insects and others) normally expected in a healthy waterway is unhealthy, reduced, or absent, and the exact cause of the problem is unknown.*

What you can do: Your state water program may have more recent information or added information not reported to EPA about your waterway. Contact your state water program to ask, or to report

anything about degraded aquatic life or possible causes that you may have observed. See EPA's CADDIS website for more information on harm to aquatic life from unknown causes.

Summary: This pollution category means that the biological community normally expected in a lake, stream or other waterway is unhealthy, much reduced, or absent, and the exact pollutant cause is not known. Over 3,200 waters are listed in this category. Degraded aquatic life associated with known causes is also a widespread problem reported under several specific pollutant names. Aquatic life includes fish, reptiles and amphibians, and a large variety of aquatic insects and other invertebrates. Normally there are enough of each of these forms of life to survive, reproduce, and serve as food for other animals. When pollution reduces or removes one form of aquatic life, this change often harms others as well. For example, a pollutant that eliminates all aquatic insects in a lake may make it unable to support fish even if the fish are not harmed by the pollutant directly. As the cause for this category is not known, it is not possible to tell whether a pollutant that has affected the fish or other life in a particular waterbody may pose a risk to human health as well. On the other hand, because this type of degradation generally involves reduction or loss of either fish or their food supply, it can impact people who make a living in the fishing industry, those who rely on fish for a source of food, and those who enjoy fishing opportunities.

CHLORINE, *used as a disinfectant and bleaching agent, is poisonous to fish and other aquatic animals at low levels. Discharges from swimming pools, storm water drains, industrial and sewage treatment facilities, and marinas can be sources of chlorine in waterways.*

What you can do: People can help reduce chlorine pollution in our waters by never dumping or rinsing off chlorine-containing disinfectants where the rinse water can wash into storm sewers or directly into a stream, lake or other waterway. Private pools should be emptied onto the ground rather than into waterways or storm drains. Read more about chlorine as a water pollutant.

Summary: Chlorine is a greenish-yellow gas that dissolves easily in water. Chlorine is not a frequently reported cause of water pollution, but over 50 waters nationwide are listed in this category. Chlorine is poisonous to fish even at very low levels. One of the most important uses of chlorine is the disinfection of drinking water to kill disease-producing bacteria. Chlorine is also used as a disinfectant in wastewater treatment plants and swimming pools, a bleaching agent in textile factories and paper mills, and is an ingredient in many laundry bleaches. Chlorine gets in our waterways from sources such as wastewater and industrial discharges and spills, urban rainfall runoff into storm water drains, and marinas. Swimming pools can be a major source of chlorinated water if they are emptied into sanitary and storm water drain systems. The storm water drain system was designed to handle runoff from rain and snow only, therefore, swimming pool water directly released into storm water drains, streets, or gutters is not treated before discharge into nearby creeks and rivers. Chlorinated waters from drinking water systems might also be released to waterways from water main breaks, leaks, and overflows. These types of releases are rarely treated before entering waterways because they happen fast and are difficult to contain. Drinking water in most towns and cities is poisonous to fish because of the chlorine it contains. Because treating municipal and industrial water supplies uses a large amount of chlorine, the excess often enters waterways where it combines with decaying material, forming other chemicals that can be cancer-causing to humans and pose a health threat to other living things.

DIOXINS, *highly toxic chemicals used in some manufacturing processes, can build up in the food chain. They may settle in sediment or on aquatic plants, then eaten and concentrated by fish, other aquatic life, wildlife, and people. Dioxins are considered likely to increase cancer risk and may harm the immune system, hormone levels, and fetal development.*

What you can do: Human exposure to dioxins largely occurs through the food we eat. To reduce your exposure to dioxins in waterways, pay attention to local fishing advisories for fish you catch and eat yourself. See more EPA and FDA information on dioxins.

Summary: Dioxins are highly toxic chemicals formed unintentionally by burning trash or leaded gasoline and as waste byproducts from manufacturing some pesticides. These chemicals can be found in fish, some waterways, and their bottom sediments. They can reach waterways through the air, by rainfall runoff and soil erosion from contaminated sites, from pulp and paper mills, and from other industrial discharges. Dioxin levels in the environment have been declining since the early seventies but are still a concern at some sites because they are long-lasting in the environment, and some dioxins are still released at low levels. Approximately 500 waters are reported as dioxin-polluted, mainly in the more industrialized states. Dioxins are considered likely to increase the risk of cancer in people and wildlife. At low doses, dioxins are linked to non-cancer effects on fetal development, immune systems, hormone levels and reproduction. Dioxins in water are found in sediments or on plants where they can be eaten and become concentrated in fish and other aquatic life. These chemicals may build up to harmful levels in fish and in the human body.

FISH CONSUMPTION ADVISORY (FISH UNSAFE TO EAT) *means that eating fish or shellfish caught from the waterway has been limited or banned, usually for certain species of fish/shellfish and for one or more chemicals, microbes or other conditions. In rivers and lakes, fish consumption advisories are usually issued because contaminants such as mercury or PCBs exceed safe limits in fish flesh; in coastal waters, shellfish harvesting may be banned due to unsafe levels of bacteria.*

What you can do: Pay attention to warnings, they are meant to protect your health. Note that most pollutants can't be seen or smelled in fish, and even if the catch appears normal the warnings still apply. EPA's website on fish advisories contains much more information than How's My Waterway on specific waters with this problem.

Summary: This reporting category means that a state has issued a warning to protect people from health risks of eating contaminated fish and shellfish caught in local waters. This advisory warning may recommend limiting or avoiding eating certain kinds of fish, fish from specific waters or from specific water types (such as "all lakes statewide"). Sometimes there are stricter advisories for pregnant women, nursing mothers, and children, all of which are more easily harmed. States also issue other guidelines to let people know that fish from some waters are safe to eat. Just 83 specific water bodies are currently listed for having contaminated fish under the polluted waters reporting process. The low number is because other affected waters have been reported under the pollutant name instead. Other state and local procedures for reporting this problem account for far more waters. The 2010 total of 4,598 advisories covers 42% of the Nation's total lake acreage and 36% of the nation's total river miles. A variety of pollutants may be responsible for warnings about eating fish, and all such warnings address risk to human health. Bans on shellfish harvest in coastal waters are often due to unsafe levels of

bacteria, which may come from sources such as sewage leaks or discharges, failing septic systems, or manure runoff. Fish advisories are also often due to unsafe levels of mercury, PCBs and other chemical pollutants that can build up in fish flesh.

FLOW ALTERATION (ABNORMAL FLOW) *refers to changes in river or stream volume caused by removing water for irrigation, water supply, and industry, and by dams, which hold and release water on a man-made cycle. Reduced flow can lower oxygen levels, raise water temperatures, cause build-up of sediment and pollutants, destroy aquatic wildlife habitat, and degrade swimming, boating, and fishing.*

What you can do: People can use less water wherever possible during droughts or when using water from waterways that already have low flow problems. See EPA websites for more information on flow alteration.

Summary: Major changes in stream or river flow are a form of pollution because they can reduce or eliminate fish survival, degrade a variety of beneficial human uses and indirectly make other pollutants more harmful. Although removing surface water for use is essential and widespread throughout the US, reporting of flow alteration as a direct cause of degradation is limited to approximately 100 waters mostly in the eastern and central states. Common causes of altered flow include water remoal for irrigation, municipal water supplies and industry. These uses of water are important, but in extreme cases they can reduce or eliminate other uses such as navigation, fishing or recreation. Some waterways with reduced flow dry up entirely as a result of withdrawals. Reduced water flow also indirectly affects many pollutants by providing less water to dilute contaminants. Lower water volumes can contribute to stagnant, warm water, buildup of mucky sediments, low oxygen and loss of fish and other aquatic life.

HABITAT ALTERATION (DEGRADED AQUATIC HABITAT) *occurs when stream channels are changed or diverted through man-made channels, artificial shorelines and stream banks replace natural ones, or native vegetation is removed from shores and banks. These actions reduce the habitat that fish and other animals need to reproduce, feed, and find shelter, and can also affect the appearance and value of waterfront property.*

What you can do: Waterfront property owners or users can reduce habitat degradation by not removing streamside vegetation or channelizing streams, not filling stream pools, wetlands or other waters, keeping natural shorelines intact, and leaving some rocks, logs or native aquatic plants as cover for fish. These actions can maintain recreational uses and appearance while avoiding unnecessary maintenance chores and costs. Read more about degraded habitat causes and effects.

Summary: Degraded habitats are areas where the conditions needed for fish and other aquatic life to feed, reproduce, find shelter, and survive have been reduced or lost. About 3,000 waters throughout the US are currently identified in this pollution category. Because damages to habitat by water flow changes or specific pollutants (such as sediment) are reported separately, this habitat degradation category mainly refers to structural changes, such as loss of pools or deep channels where fish can gather, removal of plants, logs and rocks that provide cover, or changes that make areas unsuitable for spawning. Stream straightening, channelization, filling stream pools, lining streambeds with concrete,

and replacing natural shorelines with artificial walls are common forms of man-made habitat degradation. These types of changes can harm aquatic life but do not directly pose risks to human health. However, degraded habitats often make fishing and other forms of water-based recreation undesirable, and can impact the appearance and value of waterfront property.

MERCURY *is found in many rocks, including coal. Released into the air by coal-fired power plants, it settles on land and is washed into waterways. Spills and improper treatment and disposal of mercury-containing products or wastes are among other top sources of mercury in water. Mercury can build up in fish, which then poses health risks to people and animals that eat fish.*

What you can do: People can help reduce mercury in the air and water by purchasing mercury-free products and correctly disposing of products that contain mercury. Fish consumption warnings for specific waters concerning mercury are also compiled by EPA. Read more about mercury sources, risks and health effects.

Summary: Mercury, a metal that is found in air, water and soil, is known to most people for its use in products like thermometers, switches, and some light bulbs. Mercury ranks among the top ten national causes of water pollution, with over 4,300 waters reported. Many of these reported waters are in northern states where special studies have detected large numbers of mercury-polluted lakes, including many in remote areas. As a water pollutant, mercury can build up in fish tissue, be dissolved in the water, or be deposited in bottom sediments. Mercury is found in many rocks, including coal. When coal is burned, mercury is released into the environment. Coal-burning power plants account for over half of all US man-made mercury emissions, but mercury in the air also involves worldwide sources. Burning hazardous wastes, producing chlorine, breaking mercury products, and spilling mercury, as well as improper treatment and disposal, can also release it into the environment. Mercury in the air eventually settles into water or onto land where it can be washed into water. Once deposited, certain microbes can change it into a highly toxic form that builds up in fish, shellfish and animals that eat fish. The most common way people can be exposed to mercury is by eating fish or shellfish that are contaminated with mercury. Eating fish from mercury-polluted waters should be avoided, especially by children and nursing or pregnant women. Eating mercury-contaminated fish or shellfish can affect the human nervous system and harm the brain, heart, kidneys, lungs, and immune system.

METALS OTHER THAN MERCURY *enter waterways from factories, mining, and runoff from urban areas, as well as from natural processes such as erosion of soil and rocks. At high levels, all metals such as arsenic, cadmium, chromium, copper, lead, selenium, and zinc can be toxic to aquatic animals and humans.*

What you can do: People can help by following proper disposal of metal-containing appliances and products. Read more about metals in waterways.

Summary: Metals occur in nature, although the amount occurring naturally varies according to local geology. The common metals occurring in water are arsenic, cadmium, chromium, copper, lead, nickel, selenium, zinc, and mercury, but EPA tracks mercury separately. Excess metals are the fifth most frequent reported cause of waterbody pollution, affecting over 5,900 waters nationwide. Metals in waterways can come from human activities (industrial processes, mining, and rainwater runoff from urban areas) and natural processes (mainly erosion of soil and rocks) resulting in the release of metals

into air, water, and soil. Metals at toxic levels in water are rarely due to natural causes alone. Metals on land and in soils can also infiltrate into groundwater. Disturbed soils in metals-enriched areas can wash into streams during storms. Metals in the air from industrial emissions can be deposited onto waters or land surfaces. All metals can be toxic to aquatic animals and humans at sufficiently high exposure levels. Human health problems from high exposure, such as drinking contaminated water over a prolonged period, can include damage to organs. Excess metals at toxic concentrations can affect the survival, reproduction, and behavior of aquatic animals and can result in fish kills. Additionally, toxic levels of metals can decrease a waterway's suitability for industrial and household water uses. Metals can be removed from water destined for human use, but treatment can be expensive.

NOXIOUS AQUATIC PLANTS (EXCESS AQUATIC WEEDS) *choke waterways, degrade healthy aquatic habitats, and interfere with recreational uses such as swimming, fishing, and boating. Fertilizers, leaking septic tanks, pet and livestock wastes, sewage overflows and water withdrawals can contribute to the growth of excess aquatic weeds.*

What you can do: People can help control aquatic plants in their local waters by using lawn and plant fertilizer sparingly and never before storms, regularly checking and pumping out septic tanks, never dumping plant or animal waste in a waterway, disposing of pet waste in the trash, pumping boat waste to an onshore facility, and planting native plants to prevent nutrient runoff into waterways. Read more about ways to reduce nutrient pollution that causes the harmful overgrowth of aquatic plants.

Summary: Aquatic plants include native (naturally occurs in the waterway) and non-native (brought from somewhere else), non-invasive (not harmful) and invasive (harmful) plants. Normally, most aquatic plants play important and beneficial roles in waterways. However, under certain water conditions such as warmer temperatures, too much nitrogen and phosphorus pollution, and low flow, 'noxious' growth of native or non- native plants can choke off waterways and interfere with human uses and other aquatic life. Around 220 waters have been reported in this category nationwide, and several thousand more waters are polluted by nitrogen and phosphorus (nutrient) pollution and organic enrichment, which can cause undesirable aquatic plants to become noxious. Overgrowth of both native and non-native plants can interfere with oxygen levels in the water, threaten survival of fish and other animals, make waterways unattractive, reduce property value, and degrade or prevent recreational uses including swimming, fishing, and boating. The use of fertilizers on lawns and croplands, leaky septic tanks, wastes from livestock farming, pets, untreated sewage overflow, removal of shoreline plants, and excess water withdrawal all can create favorable conditions for harmful overgrowth of aquatic plants.

NUISANCE EXOTIC SPECIES (NUISANCE PLANTS OR ANIMALS, FOREIGN), *often called invasive species, are plants, animals, fish, or microbes that are not native to the region and cause harm to native species, to recreation and other uses of the waterway, and/or to human health. In general, invasive species spread vigorously and enter waterways by many means such as accidental or intentional releases and attachment to boats and other recreational equipment.*

What you can do: People can help prevent the spread of aquatic invasive species by never dumping aquarium fish, plants or water into local waters, inspecting and thoroughly cleaning boats, trailers, and recreational equipment before use and after use, allowing watercraft to dry completely before

launching into another body of water, and never releasing live baitfish or other bait. Learn more about waterways degraded by non-native, invasive species and how to help. EXIT Disclaimer

Summary: Nuisance species (also called invasive species) are non-native plants, animals, or microbes whose introduction to a waterway can be harmful to the environment, economy, or human health. Invasive species are one of the largest threats to marine and fresh waters. They can take over waterways from desirable native plants and animals, degrade water quality and fish habitat, and reduce water availability. In turn, they can cause economic losses by reducing recreational and commercial activities such as sport and commercial fishing, boating, shipping, swimming, and shellfish consumption. Invasive species also can decrease aesthetics and property value, and clog industrial and municipal water pipes. The costs to control and eradicate these species in the U.S. alone amount to more than $137 billion annually. Common sources of aquatic invasive species introduction include ballast water from ships, boat hull fouling, aquaculture escapes, and other accidental and/or intentional releases. Even though invasive species affect many waterways, only 28 waters are currently listed under this specific pollution reporting category. The reason is that many waters polluted by nuisance species are listed in categories such as excess sediment or low oxygen where an aquatic invasive species is the source of the problem.

NUISANCE NATIVE SPECIES (NUISANCE PLANTS OR ANIMALS, NATIVE) *includes aquatic plants and animals that are native to the region (not brought in from elsewhere) but have become too crowded in the waterway due to other pollution. Overgrowth can interfere with oxygen levels in the water, threaten survival of fish and other animals, make waterways unattractive, reduce property value, and degrade or prevent recreational uses including swimming, fishing, and boating.*

What you can do: People can help control aquatic plants in their local waters by using lawn and plant fertilizer sparingly and never before storms, regularly checking and pumping out septic tanks, never dumping plant or animal waste in a waterway, disposing of pet waste in the trash, pumping boat waste to an onshore facility, and planting native plants to prevent nutrient runoff into waterways. Read more about ways to reduce nutrient pollution that causes the harmful overgrowth of aquatic plants.

Summary: Very few waters have been reported in this category nationwide, although other reporting categories exist with higher numbers for nuisance non-native plants and algae overgrowth. Normally, most species of native aquatic plants play important and beneficial roles in waterways. However, under certain water conditions such as warmer temperatures, too much nitrogen and phosphorus pollution, and low flow, abnormal growth of a few types of native plants can choke off waterways and interfere with human uses and other aquatic life. Under these same conditions, non-native plants can become a problem as well. Overgrowth of both native and non-native plants can interfere with oxygen levels in the water, threaten survival of fish and other animals, make waterways unattractive, reduce property value, and degrade or prevent recreational uses including swimming, fishing, and boating. The use of fertilizers on lawns and croplands, septic tank failure, wastes from livestock farming and pets, untreated sewage overflow, removal of shoreline plants, and excess water withdrawal all can create favorable conditions for harmful overgrowth.

NUTRIENTS (NITROGEN AND PHOSPHORUS) *in excessive amounts can cause aquatic plants to grow too fast, choking waterways, causing potentially harmful algae blooms, and creating low oxygen conditions that can harm fish and other aquatic life.*

What you can do: People can help reduce nitrogen and phosphorus pollution in their local waters by using lawn and plant fertilizer sparingly and never before storms, regularly checking and pumping our septic tanks, never dumping plant or animal waste in a waterway, disposing of pet waste in the trash, pumping boat waste to an onshore facility, and planting native plants to prevent nutrient runoff into waterways. Read more about nitrogen and phosphorus pollution and learn more about what you can do to help reduce it. Technical details on nitrogen and phosphorus pollution can be found here.

Summary: Nitrogen and phosphorus (also called nutrients) are natural elements in the environment that are essential for plant and animal growth in normal amounts but are harmful in excess – too much of a good thing. These are among the top water pollutants nationally, degrading over 100,000 river and stream miles and over 3.5 million acres of lakes, reservoirs and ponds. About 6,000 nutrient-polluted waterbodies have been reported throughout the US. Most nutrient pollution comes from runoff or discharges from fertilizing lawns and croplands, municipal waste treatment systems, and animal wastes from livestock farming. Excess nitrogen or phosphorus can cause too much aquatic plant growth and algae blooms, sometimes choking off waterways and causing toxic or oxygen-poor conditions that can kill fish and other aquatic life. Nitrogen and phosphorus pollution can be harmful to human health if the affected waterway is used for swimming or drinking water. Nitrates in drinking water wells have been linked to the fatal "blue baby syndrome." These pollutants can also harm local economies through increased drinking water treatment costs, poor fish and shellfish harvests, less income from reduced recreational tourism, and potentially reduced property values on polluted waterways.

OIL AND GREASE *includes fuel oil, gasoline, vegetable oil, and animal fats. Oils generally enter waterways through spills, leaks, and improper disposal, and can be toxic to plants and animals even in small amounts.*

What you can do: People can help reduce oil and grease pollution by always disposing of car oil and paints properly and never in storm sewers and drains, cleaning up spilled oil and grease with absorbent towels instead of hosing them into the street where they can eventually reach local waterways, and fixing oil leaks from vehicles right away. Read more about things you can do to prevent urban runoff leading to oil and grease pollution.

Summary: Oil and grease pollutants (oils) include petroleum (fuel oil, diesel oil, and gasoline) and non petroleum (vegetable oil and animal fats) oils. Oils are almost everywhere in small amounts, but they are a reported cause of water pollution in about 150 waters nationwide. This pollutant tends to enter waterways as a result of leaks and spills occurring on land and on the water. Although large, major spills tend to be highly publicized and can do significant damage to waterways, small unreported spills also damage local waters and are more common. Oil and grease pollution affecting inland waters is often the sum total effect of many car/truck oil leaks, small unreported spills, or improper disposal of used oil that makes its way into storm drains. Other sources of spills and leaks can include oil production onshore and offshore, industrial food production facilities, fueling stations (marine and land), boats, and jet skis. Although heavier oils may sink and build up around rocks and sediments, most oils tend to float and spread on the water surface, creating a slick. Wind, water currents, and warmer waters can cause slicks to spread. Without much water movement, oils tend to collect in one spot and remain for long periods of time. Even in small amounts, oil can be toxic to plants and animals that live on or around the

water surface and those that live under water, resulting in smothering or toxic effects. Spilled oil can also damage parts of the food web, contaminating fish and plants that we eat and water used for drinking.

ORGANIC ENRICHMENT/OXYGEN DEPLETION (LOW OXYGEN) *levels in water can occur naturally for short periods, but when they are extreme or long-lasting, they can sicken and even kill fish and other aquatic animals. Sewage wastewater, leaking septic tanks, farm and feedlot runoff, and runoff from city streets contain organic materials that decompose and use up oxygen in water; higher water temperature also lowers oxygen levels.*

What you can do: People can help avoid low dissolved oxygen problems in their local waters by never dumping plant or animal waste in a waterway, applying the correct amount of fertilizer on lawns and never before storms, disposing of pet waste in the trash, pumping out septic tanks regularly, and pumping boat waste to an onshore facility. Read more about dissolved oxygen pollution and what you can do to reduce nutrient pollution that results in organic enrichment and low dissolved oxygen.

Summary: Dissolved oxygen in the water is essential for healthy waterways. Aquatic plants consume oxygen at night even in healthy waters, so oxygen levels in the water can change naturally. Severe depletion of oxygen, however, is usually due to human activities that increase the amount of plant parts, chemicals or animal and human waste in the water. Prolonged periods of low dissolved oxygen are harmful to most aquatic life and can cause fish kills and large dead zones (areas that can't support aquatic life). Low dissolved oxygen and decay can cause foul smells and make waterfront properties and recreation unattractive. When excess organic matter enters the water and decays, it depletes the oxygen below levels that fish and other aquatic life forms need to survive. Some types of chemical pollutants also decrease oxygen in water and have similar effects. Runoff of chemical and manure-based fertilizer applied to lawns and croplands, septic or untreated sewage overflow, animal wastes from livestock farming and pets, and industrial waste such as discharges from pulp and paper mills can cause low oxygen. Reservoirs and activities that involve straightening streams can also cause oxygen-poor waters because they mix the air and water less than normal streamflow and decrease aeration. Prolonged high temperatures can also decrease oxygen since warm water cannot hold as much oxygen as cold water. Around 6,000 waters have been reported in this category nationwide, making this the third most common reporting category, and several thousand more waters with nitrogen and phosphorus pollution or high temperature also affect dissolved oxygen in waters.

OTHER CAUSE *is a 'miscellaneous' reporting category used for dissolved gases, floating debris and foam, leachate, stormwater pollutants, and many other uncommon causes lumped together.*

What you can do: Your state water program may have more detailed information not reported to EPA about pollution causes. Contact your state water program with questions or to report what you have observed that may involve pollution causes.

Summary: This reporting category is not commonly used, and includes about 200 waters nationwide from recent reporting. Waters in this 'miscellaneous' category represent a wide variety of types of problems. Some examples include dissolved gases, floating debris and foam, leachate, stormwater pollutants, and many other causes. Due to the variety of causes and sources, it is difficult to generalize

about this category's potential effects on human health and beneficial uses or environmental impacts, or provide links for additional detailed information.

PATHOGENS (BACTERIA AND OTHER MICROBES) *are potentially disease-causing organisms from human or animal wastes that enter waters through septic tank leaks or sewage discharges, farm and feedlot manure runoff after rain, boat discharges, and pet and wildlife waste. People can become ill by eating contaminated fish or shellfish or swimming in waters with high levels of these microbes.*

What you can do: People can help reduce pathogen contamination by never dumping animal or boat waste in a waterway, fixing leaky septic tanks, picking up pet waste, and avoiding manure application close to shorelines or drainage ditches. Read more about pathogens in waterways and drinking water and health risks from pathogens.

Summary: Disease-causing bacteria and other microbes (viruses and protozoa) are called pathogens, and they usually come from human or animal waste. They are the most commonly reported cause of water pollution nationwide, with over 10,300 waters identified. These microbes enter US waterways from both man-made and natural sources, and can affect human and animal health as well as several beneficial uses. They reach the water directly in urban and suburban areas from wastewater treatment plants, sewer overflows, failing sewer lines, slaughterhouses and meat processing facilities; tanning, textile, and pulp and paper factories; fish and shellfish processing facilities; sewage dumped overboard from recreational boats; and pet waste, litter and garbage. Rural sources include livestock manure from barnyards, pastures, rangelands, feedlots, unfenced farm animals in streams, improper manure or sewage land application, poorly maintained manure storage, and wildlife sources such as geese, beaver and deer. The amount of bacteria and other microbes present, and thus the health risks they represent, can change rapidly due to factors such as rainfall and runoff from the sources mentioned above. Serious but rarely life-threatening illnesses are caused mainly by swallowing pathogen-contaminated water during swimming or other recreation, but can also come from skin contact with the water or eating contaminated fish or shellfish. Livestock, pet, and wildlife illnesses can also occur. Besides causing illnesses, pathogens in waterways can cause significant economic losses due to beach closures, swimming and boating bans, and closures of shellfish harvest beds. When present in raw drinking water sources, they can be treated but require advanced and expensive methods to disinfect and filter the water supply.

PESTICIDES *such as herbicides and insecticides include a variety of toxic chemicals used to kill unwanted pests or weeds. In water, pesticides can affect the health of aquatic insects, fish, plants, and animals who are exposed through feeding or contact.*

What you can do: People can reduce pesticide pollution in waterways by always using insecticides and herbicides in proper doses, well away from waters or drainage ditches, only on still days, and disposing of waste properly. See more information on pesticide human health effects, insecticide effects on waterways, or herbicide effects on waterways.

Summary: Pesticides (including insecticides, fungicides and herbicides) are a broad variety of chemicals used to kill unwanted pests or plant life. About 1,000 waters throughout the US are currently reported as polluted by pesticides. Although pesticides are mainly used around homes, forestry, and agriculture,

they can easily enter waters through direct application, drift from airborne applications, stormwater or irrigation runoff, discharge from industries, or wastewater treatment plants. Timing and amount of pesticide used, rainfall and wind after use, and how fast the pesticide degrades all affect how much of it may reach the water. The potential human health effects of pesticides depend on the type of pesticide and amount of exposure, but can include nerve damage, hormonal effects, skin or eye irritation, or cancer-causing or reproductive effects. However, in many cases the amount of pesticide to which people are likely to be exposed is too small to pose a risk. Insecticide and herbicide effects on waters can be significant. Aquatic insects may be especially susceptible to insecticides, affecting a main food supply for fish. Fish themselves also can be killed or affected by slowed growth, less disease resistance, and poor reproduction. Death of aquatic plants from herbicides can remove food sources and cover for aquatic life, reduce oxygen and water quality, and degrade fish habitat.

PH/ACIDITY/CAUSTIC CONDITIONS (ACIDITY) *outside a certain range can sicken or kill fish and other aquatic life. Highly acidic or alkaline water can also release pollutants from sediments that can further harm aquatic life. Acidity in waterways is influenced by rock and soils, as well as human sources such as industrial and car emissions, mining, and agricultural runoff.*

What you can do: People can help reduce pH problems by applying the correct amount of fertilizer on lawns (and never before storms), properly disposing of chemicals such as household cleaners, and never dumping any of the above into ditches, waterways and storm drains. Read more about pH, and what you can do to help reduce acid rain.

Summary: The health and survival of aquatic plants and animals depends heavily on pH, which is a measurement of how acidic or basic the water is. Think of acid and base as two extremes, with neutral in the middle; a pH toward either extreme is generally harder for aquatic life to survive. Most aquatic plants and animals under those extreme conditions have reduced ability to grow, reproduce, and survive. Low pH (acidic) can cause toxic metals such as aluminum and copper to dissolve into the water from bottom sediments. High pH (basic or alkaline conditions) can increase the toxic form of ammonia, which can further harm fish and other aquatic life. Natural sources that influence acidity in waterways are the surrounding rock and soils, and processes such as decay of plants. Human activities that can result in acidity include agriculture (animal feedlots), urbanization and industry (emissions from vehicles and coal-fired power plants leading to acid rain and ocean acidification), and mining (acid mine drainage). Although human activities commonly result in more acidic conditions, high alkaline conditions can occur by means of stormwater runoff from sources associated with agriculture (lime-rich fertilizers) and urbanization (asphalt roads), wastewater discharges and leakage from sources associated with industry (e.g., soap manufacturing plants), and mining (oil and gas brine mining wastes). Around 4,000 waters have been reported as polluted by pH problems, making this the 8th most common reporting category.

POLYCHLORINATED BIPHENYLS (PCBs) *are a toxic mixture of industrial chemicals which, although banned since the 1970s, are long-lasting in fish tissue and in the bottom sediments of rivers and lakes. PCBs in fish that are eaten by humans and wildlife can build up and may have cancer-causing and other health effects. PCB contamination has caused many fishing bans and warnings.*

What you can do: Your state water program may have more information about PCBs not reported to EPA. Contact your state water program with questions. See EPA websites for basic PCB information and PCB health effects.

Summary: PCBs, or polychlorinated biphenyls, are a toxic mixture of chlorinated chemicals that were banned in the late 1970s but are still a common pollutant because they build up in fish flesh and are long-lasting in the bottom sediments of rivers and lakes. Over 4,500 water bodies are currently listed in the PCB-polluted category, making this the sixth-highest water pollution cause. PCBs have reached waterways worldwide by direct dumping, leakage from landfills not designed to handle hazardous waste, and through the air after burning PCB-containing waste. Originally PCBs were widely used in industry, particularly as coolants and lubricants in transformers and other electrical equipment. PCBs have been shown to cause cancer in animals. Studies have also provided evidence of potential cancer-causing effects in humans. Non-cancer health effects on the immune system, reproductive system, and nervous system in animals have been documented. PCBs are also related to deformities in birds and heart effects in young fish. PCB risks to human health occur when PCBs build up through eating PCB-contaminated fish and other sources. Other negative effects on people include recreational and commercial fishing bans at numerous PCB-contaminated lakes and rivers and the related economic impacts over the past 30 years.

RADIATION *can enter waterways through eroding or dissolving underground deposits of radioactive metals such as uranium, from the air due to accidental or intentional release, in seepage from improper disposal sites, in mining runoff or dumped mine tailings, or from industrial activities. It can become a health concern when radioactive materials become concentrated in waterways.*

What you can do: Read more about radiation and US waterways.

Summary: Although quantities that pose a health risk are uncommon and localized, radiation can be a water pollutant in some US waterways. 32 polluted waters currently occupy this reporting category. Radioactive atoms, known as "radionuclides," are a water pollutant that comes originally from underground deposits of radium, uranium and other radioactive metals. Radioactive materials can enter water by being deposited in surface water from the air, by entering ground water or surface water from the ground through erosion, seepage, or human activities such as mining, farming, storm water, and industrial activities, or by dissolving from underground mineral deposits as water flows through them. Health becomes a concern when radionuclides become concentrated in bodies of water due to natural occurrences, accidental releases of radioactivity, or improper disposal practices. The primary environmental and human health risks from radiation involve cancer, but the degree of risk varies with how much radiation is involved over how long a time period.

SALINITY/TOTAL DISSOLVED SOLIDS/CHLORIDES/SULFATES (SALTS) *are minerals that dissolve in water; they can be toxic to freshwater plants and animals and make water unusable for drinking, irrigation, and livestock. Water withdrawals, road de-icing, human and industrial wastewater, fertilizer applications, mining and oil or gas drilling, and repeated use of irrigation water contribute to high levels of salts*.

What you can do: People can help by minimizing the use of de-icing salts where they may be washed off into waterways, storm drains and ditches. Please see more information on the sources and effects of salts on our waterways.

Summary: Salts are minerals that dissolve in water. Common table salt is a familiar example that consists of sodium and chloride, but salts can also consist of other minerals such as calcium magnesium, sulfate, bicarbonate, and potassium. Dissolved salts are essential to life in our waters when in small quantities, but too much is harmful to freshwater aquatic life and many human uses. More than 1000 normally fresh water bodies across the country have been listed as polluted because they contain too much salt. Most freshwater plants and animals tolerate only very low amounts of salts, and can sicken or die when these ranges are exceeded. Although salts occur naturally, human activities can increase salts to beyond the range tolerated by freshwater aquatic life. At higher salt levels, water becomes unusable for drinking, crop irrigation, livestock watering, and manufacturing. Some of the sources and activities that increase the salts in streams, lakes, groundwater and other waters include disposal of human and industrial wastewater, fertilizer and lime application, irrigation, mining and oil production, weathering of cement in urban areas, salt-water intrusion into drinking water supplies in arid areas and along the coasts, and de-icing treatment of roads and other surfaces during the winter. Once in a waterway, excess salt is very difficult to remove. Preventing salt from entering water in the first place is the best management strategy.

***SEDIMENT** is a problem when rain washes soil into waterways from fields, construction sites, yards, logging areas, city streets and other disturbed areas. Sediment can make water murky, hurt the health and habitats of fish and other aquatic animals, interfere with uses like fishing and swimming, and carry other pollutants sometimes including toxic chemicals.*

What you can do: People can help reduce sediment pollution by limiting soil erosion in any way possible, including not removing native plants from stream edges, not disturbing soil near ditches or waterways, and routing rainwater to areas where it can soak in rather than directly dump into a lake, stream or sewer system. Read more about sediment effects on waterways and ways to help control sediment problems.

Summary: Sediment is material eroded from rocks or soil and then transported and deposited in water. Sediment in the proper quantity is a natural part of the banks and bottom of lakes, streams and other waterways, but it becomes a problem when too much fine sediment enters the water or when it is contaminated by other pollutants. Excess fine sediment is one of the most common forms of pollution, reported in over 6,000 water bodies from all parts of the US. These waters most often suffer from excessive suspended sediment in the water or too much deposited fine sediment on the bottom. Too little sediment below dams sometimes causes streams to scour their channels and destroy fish habitat. Sediment problems happen when rain washes silt and other soil particles off of plowed fields, construction sites, logging sites, urban areas, and strip-mined lands into waterbodies. The sediment may clog and damage fish gills or suffocate eggs and aquatic insects on the bottom. Suspended silt may interfere with recreational activities like boating, fishing or swimming and degrade the beauty of waterways by reducing water clarity. Although sediment itself is generally harmless to human health or safety, indirect environmental or health risks can happen when nitrogen and phosphorus

pollution and a variety of toxic chemicals attach to sediment particles on land and ride the particles into surface waters.

TASTE, COLOR AND ODOR *problems may indicate that pollutants are present; however, these problems are of concern mainly because they affect uses of waterways, such as swimming, drinking water supply, or aesthetic enjoyment.*

What you can do: Never dispose of any kind of waste into or close to any waterway. Learn more about taste, color and odor in drinking water.

Summary: This category of waterways may imply that water pollutants are present, but it is based mainly on the undesirable sensations they cause rather than for actual harm to human or environmental health. Although an unpleasant taste, color or odor may not be harmful to people or the environment, it can have a powerful effect on whether a waterway is acceptable by a community for many beneficial uses. Odor and taste, which can be caused by a wide variety of dissolved substances, are useful indicators of water quality even though odor-free water is not necessarily safe to drink. Color may be indicative of dissolved plant material or the presence of dissolved metals. Over 100 waters nationwide are currently listed for taste, color or odor problems, but only a small minority of states uses this reporting category. Most state water quality standards say generally that lakes, streams and other waters must be free from objectionable odors, tastes or colors, regardless of their use. But when the waterway is also a drinking water source, these characteristics become much more important because unpleasant levels can cause a community to reject the source as drinking water or require additional, expensive drinking water treatment to remove tastes, colors or odors. Further, unpleasant colors or odors in recreational waterways can lead people to reduce or stop their recreational uses of these areas, leading to local economic losses.

TEMPERATURE: *Many fish and other aquatic animals are sensitive to changes in water temperature and require a certain temperature range to survive. If water temperature goes outside that range for too long, they can sicken or die.*

What you can do: People can help avoid water temperature problems by not removing shade trees and shrubs from streambanks, using less water during droughts, and directing rainwater on pavement to soak into the ground instead of running into streams, lakes, or sewer systems. See more information on water temperature.

Summary: Abnormally high water temperature impacts aquatic life in many streams, lakes and other waters nationwide. About 3,000 waters have been reported as degraded by high temperature, mostly in the Northwest and the Northeast, due to concerns over salmon and trout survival. Waters can become too warm for fish and other life due to rain running off hot pavement, warmer water discharges from industry or agriculture, increased sunlight from streambank vegetation removal, and major water withdrawals in summer, leaving less water that heats more rapidly in the sun. High water temperatures can harm or kill fish and other life mainly by reducing the oxygen in the water or by raising temperatures above their survival limits. Warmer waters can also increase toxicity of pollutants, cause faster growth of undesirable algae blooms, and increase the spread of diseases in fish. Although high water temperature

does not directly affect human health, it can speed up the growth of waterborne bacteria or toxic algae that can harm people or their pets if swallowed or contacted. Elevated temperature also directly degrades valuable uses such as recreational fishing, boating, and commercial salmon fishing.

TOTAL TOXICS *include a large number of harmful, man-made substances such as solvents, pesticides, fungicides, dioxins, PCBs, and furans. They enter waterways through improper application and disposal, runoff, spills, auto exhaust, and burning of chemical wastes. These chemicals are toxic to animals and people.*

What you can do: People can help eliminate toxics in waterways by never rinsing out contaminated containers or dumping directly into waterways or storm sewers. Also never flush down the toilet anything known to be poisonous, such as paints, paint strippers, other solvents, cleansers and disinfectants, prescription drugs, and automotive products. Read more about toxic chemical effects in waters and what you can do to help reduce toxic chemicals in our waterways.

Summary: Total toxics is a term used when a mix of harmful chemical pollutants occurs in a waterway. Roughly 300 waters nationwide are in this reporting category, which is used when the exact types of chemicals in the water are not specified. Toxics in water or contaminated sediment may have come from industrial activities, wastewater treatment plants, landfills or hazardous waste sites. The potential for toxics to harm living things is dependent on the type and amount of the chemicals and how long a living thing has been exposed to them. Toxic chemicals in water can harm aquatic plants and animals by decreasing reproduction, increasing disease, and in some cases causing death. Toxic chemicals in higher amounts and over time generally can harm people's immune, reproductive, and nervous systems, and in some instances are cancer-causing.

TOXIC INORGANICS *refers to a wide range of pollutants including metals, fire retardants, cyanide, and perchlorate (used in rocket fuel) that are poisonous to aquatic life and people. Industrial or wastewater discharges, mining, landfills, and air deposition of car exhaust and coal-fired power plant emissions can contribute to high levels of toxic inorganic chemicals in waterways.*

What you can do: People can help eliminate toxics in waterways by never rinsing out contaminated containers or dumping directly into waterways or storm sewers. Also never flush down the toilet anything known to be poisonous, such as paints, paint strippers, other solvents, cleansers and disinfectants, prescription drugs, and automotive products. Read more about toxic chemical effects in waters and what you can do to help reduce toxic chemicals in our waterways.

Summary: Toxic inorganics are human-made or naturally occurring chemicals that can harm the health of aquatic life and people if exposed at high enough levels. Toxic inorganic pollutants include a wide range of chemicals from a wide array of sources. The most common toxic inorganic water pollutants are reported separately in their own categories, including mercury (over 4,000 waters reported), and other metals (around 6,000 waters reported). Around 360 other waters have been reported under the category of toxic inorganics, including antimony (used as a fire retardant in textiles and plastics), fluoride (added to drinking water to promote dental health), ozone (used to treat water to kill bacteria and viruses), cyanide (used in metal treatment), and perchlorate (used in rocket fuel). Human activities are

usually responsible for introducing toxic concentrations of inorganic chemicals to waterways, including direct discharges from industrial or wastewater treatment plants, rain runoff and leakage from agricultural fields, mining operations, landfills, and rocket fuel manufacturing sites, and air deposition from car exhaust and coal-fired power plants.

TOXIC ORGANICS *are harmful, man-made chemicals that all contain carbon. They can build up in animal and fish tissues and sediments or get into drinking water supplies, posing potential long-term health risks.*

What you can do: People can help eliminate toxics in waterways by never rinsing out contaminated containers or dumping directly into waterways or storm sewers. Also never flush down the toilet anything known to be poisonous, such as paints, paint strippers, other solvents, cleansers and disinfectants, prescription drugs, and automotive products. Read more about toxic chemical effects in waters and what you can do to help reduce toxic chemicals in our waterways.

Summary: Toxic organic chemicals are harmful, man-made chemicals containing carbon. These often remain in the environment for long periods and can accumulate in animal and fish tissues and sediments. They also can get into drinking water supplies, posing potential long-term health risks to humans. Toxic organic chemicals are the reported cause of water pollution in over 280 waters nationwide. These pollutants include a large number of chemicals such as solvents, pesticides, dioxins, PCBs, furans, and other nitrogen compounds. Common sources include wood preservatives, antifreeze, dry cleaning chemicals, cleansers, and a variety of other chemical products. Two important sources of toxic organic chemicals in water are improper disposal of industrial and household wastes and runoff of pesticides. Excessive application of insecticides, herbicides, fungicides, and rodenticides, or application of any of these shortly before a storm, can result in toxic chemicals being carried by stormwater runoff from agricultural lands, construction sites, parks, golf courses, and residential lawns to receiving waters. Other organic pollutants come from auto exhaust and from burning municipal and chemical wastes. Organic pollutants can build up in aquatic animals and increase in concentration. These substances can be toxic to all forms of life, and are known to cause cancer in animals. For humans, some of them are suspected to cause cancer and are also known to be harmful to immune, reproductive, nervous, and hormone systems.

TRASH *consists of litter, debris, and other types of discarded solid waste. Trash can be contaminated with toxins or bacteria, and it harms fish and wildlife that eat it or become entangled in it. In areas where people swim or wade, it also poses a human health and safety threat.*

What you can do: Never use waterways or their sloping banks as a place to dump garbage or litter of any amount. People can help by properly disposing of trash, not littering in or near waterways, preventing trash from being blown away, and picking up visible trash in and near waterways. Learn more about trash in fresh waters, marine debris, and case studies in waterway trash control.

Summary: Trash consists of litter, debris, rubbish, refuse and other types of solid waste discarded by people. Trash in waterways is common and unsightly, but not usually enough to be the main cause for reporting a waterway as polluted. In fact, trash is the main reporting category for 59 polluted waters

nationwide. Litter left on sidewalks, streets, yards or other open areas may be carried by rainwater to storm drains that discharge into waterways. Trash can also be carried to waters from nearby areas by wind or rainwater runoff. Also of concern are trash "hotspots" where it piles up from illegal dumping and littering, such as on steep streambanks below a roadside pull-off. What happens to trash in waterways depends on trash size, ability to float, and rate of deterioration. Marine trash or debris, which degrades ocean beaches, comes from ocean dumping and beach litter. Once trash enters a waterway, it can float (used plastic food containers, wrappers and cans), sink (glass containers, cigarettes), or become suspended underwater (plastic grocery bags), and degrade the habitat and health of aquatic plants and animals. Floating litter in water may be contaminated with toxic chemicals and bacteria, is unattractive to look at, and can harm aquatic animals and birds if they eat trash or become entangled. Trash that sinks can contribute to sediment contamination, and large trash items such as discarded appliances can result in stream erosion or contamination. Trash in waters can threaten the health or safety of people who use them for wading or swimming. Of particular concern are the bacteria and viruses associated with diapers, medical waste such as needles, and human or pet waste. Some trash items such as containers or tires can hold still water that grows mosquitos. Litter degrades the appearance and quality of waterways that provide recreation, drinking water, and numerous other benefits to society.

TURBIDITY (MURKY WATER) *refers to water that is cloudy, muddy or opaque (turbid) because of suspended soil particles, algae, microbes, or organic matter. These tiny particles can absorb heat and raise water temperatures, reduce oxygen for aquatic animals, reduce native aquatic plant growth, clog fish gills and smother fish eggs and aquatic insects.*

What you can do: Waterfront property owners or users can reduce turbidity by not removing streamside vegetation or channelizing streams, not filling wetlands or other waters, keeping natural shorelines intact, leaving some rocks, logs or native aquatic plants as cover for fish, and routing rainwater runoff to areas where it can soak in rather than directly dump into a lake, stream or sewer system. See also EPA information on why turbidity is important and on reducing and controlling turbidity in drinking water.

Summary: Turbidity is a measure of how 'murky' the water is, reported as a pollution cause for over 3,000 waters nationwide. Tiny particles of suspended matter or impurities can make water cloudy, muddy or opaque (turbid). Materials that cause water to be turbid may include clay, silt, fine organic matter, and microscopic life such as algae. The primary source of turbidity is rainwater runoff from disturbed or eroding land. Additional sources may include urban waste discharges, as well as particles from the decay of plant materials. High turbidity can reduce light penetration and degrade or eliminate aquatic plants in lakes and estuaries, leaving poorer shelter, nurseries, and food for fish and other aquatic animals. Loss of aquatic plants then allows wind and waves to stir up more cloudiness, which can make waters unattractive for recreational use. Suspended particles also increase temperature, reduce oxygen in water, clog fish gills and reduce survival of fish eggs. Although turbidity is not a direct cause of human health risk, other pollutants such as metals and bacteria may attach to suspended particles. If not controlled, turbidity can promote growth of bacteria, leading to waterborne diseases such as intestinal illnesses after swimming. Numerous studies show a strong relationship between reduction of turbidity and reduction of some disease-related microbes.

Exhibit F

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Water chemistry and nutrient release during the resuspension of FeS-rich sediments in a eutrophic estuarine system

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ABSTRACT

The objective of this study was to investigate the impact of resuspending FeS-rich benthic sediment on estuarine water chemistry. To address this objective, we conducted (1) a series of laboratory-based sediment resuspension experiments and (2) also monitored changes in surface water composition during field-based sediment resuspension events that were caused by dredging activities in the Peel–Harvey Estuary, Western Australia. Our laboratory resuspension experiments showed that the resuspension of FeS-rich sediments rapidly deoxygenated estuarine water. In contrast, dredging activities in the field did not noticeably lower O_2 concentrations in adjacent surface water. Additionally, while FeS oxidation in the laboratory resuspensions caused measurable decreases in pH, the field pH was unaffected by the dredging event and dissolved trace metal concentrations remained very low throughout the monitoring period. Dissolved ammonium (NH_4^+) and inorganic phosphorus (PO_4–P) were released into the water column during the resuspension of sediments in both the field and laboratory. Following its initial release, PO_4–P was rapidly removed from solution in the laboratory-based (<1 h) and field-based (<100 m from sediment disposal point) investigations. In comparison to PO_4–P, NH_4^+ release was observed to be more prolonged over the 2-week period of the laboratory resuspension experiments. However, our field-based observations revealed that elevated NH_4^+ concentrations were localised to <100 m from the sediment disposal point. This study demonstrates that alongside the emphasis on acidification, deoxygenation and metal release during FeS resuspension, it is important to consider the possibility of nutrient release from disturbed sediments in eutrophic estuaries.

1. Introduction

Many shallow estuarine environments often require dredging to maintain access to boating channels and residential canals. In such environments, dredging activities may contribute to the resuspension of anoxic benthic sediment into overlying oxic surface waters. It has been documented that resuspension of benthic sediments can rapidly deoxygenate surface waters (Sullivan et al., 2002; Fyfe et al., 2007). In addition, sediment resuspension can trigger the oxidation of iron monosulfide (FeS) minerals and cause pH-dependent release of FeS-associated trace metals (Burton et al., 2006a; Bush et al., 2004; Simpson et al., 1998; Johnston et al., 2003; Wong et al., 2010).

In marine and estuarine sediments, sulfate (SO_4^{2-}) reduction is often a dominant mechanism for the mineralisation of organic matter (Capone and Kiene, 1988; Jørgensen, 1982; Morgan et al., 2012). In addition to contributing to sedimentary FeS accumulation, mineralisation of organic matter and related sulfur (S) cycling can also induce enrichments in porewater ammonium (NH_4^+) and phosphate (PO_4–P, which includes $\sum PO_4^{3-}$, HPO_4^{2-}, $H_2PO_4^-$) (Caraco et al., 1989; Smolders et al., 2003, 2006). Despite this, nutrient release is rarely regarded as an important environmental concern in relation to the resuspension of FeS- and organic-rich sediments.

The objective of the current study was to investigate the impact of estuarine sediment resuspension on surface water chemistry and nutrient concentrations. This study focuses on a eutrophic estuary in Western Australia that is known to contain benthic sediments rich in FeS (Morgan et al., 2012). We addressed the objective by employing (1) a laboratory-based (closed-system) set of sediment resuspension experiments and (2) a field-based (open-system) monitoring programme examining surface water composition during large-scale sediment disturbance caused by dredging activities.

2. Materials and methods

2.1. Study site description

This study investigates the resuspension of sediments from the frequently-dredged South Yunderup channel in the Peel–Harvey Estuarine System (PHES) (Fig. 1). The PHES is located on the Swan coastal plain approximately 80 km south of Perth, Western Australia

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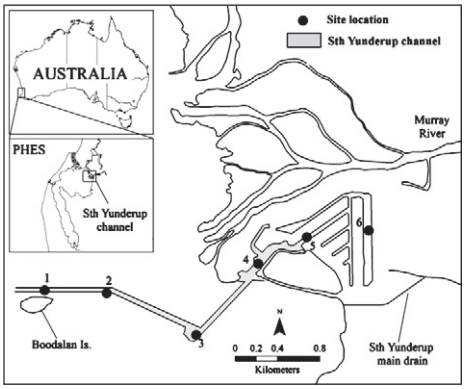


Site	Latitude ("S)	Longitude ("E)
1	32°35'30.60	115°45'10.09
2	32°35'31.06	115°45'30.72
3	32°35'41.16	115°45'52.35
4	32°35'.29.21	115°46'07.89
5	32°35'20.43	115°46'25.37
6	32°35'15.86	115°46'46.72

Fig. 1. A map of the South Yunderup channel, with Sites 1–6 representing the sediment collection points for analysis of general sediment properties (see Section 2.3). Sediment was collected from Site 4 for laboratory resuspension and deoxygenation experiments.

(Fig. 1). The system is of high recreational use and frequent dredging is required to maintain access to the boating navigation channels and residential canals (Brearley, 2005). There is ~10% flushing of estuarine water to the Indian Ocean daily, with flood tides of 2.17 m/s and ebb tides of 2.07 m/s (Brearley, 2005). Additional details of the PHES can be found in Morgan et al. (2012), including information on the characteristics of the FeS-rich benthic sediments.

The dredge event monitored in this study operated between Sites 1 and 5 (Fig. 1) of the South Yunderup channel. Intact sediment cores were collected from the South Yunderup channel (Sites 1 to 6) with the use of a polypropylene coring device for geochemical characterisation of the undisturbed sediments (See Section 2.2). At Site 2, three cores were collected within a 1 m × 1 m area to investigate field variability. At all other sites only a single core was collected. The length of cores varied between 4 cm and 25 cm depending on our ability to penetrate the sediment with the coring device. Core lengths were much shallower than the dredge depth (which was >1 m) and as such the profiles provide only an indication of the geochemistry of the bulk sediment that was disturbed in the field. Each core was extruded and rapidly (<2 min) sectioned into 2 cm (for the 0–10 cm depths) or 5 cm (for the 10–25 cm depth) segments. These segments were then placed into thick plastic zip-sealed bags with all air excluded. Each sample was homogenised by manual manipulation of the sample bag for approximately 1 min. One corner of the bag was then cut away and the sample was squeezed into 50 mL centrifuge tubes and sealed, with no bubbles or headspace, by a gas tight screw cap. These centrifuge tubes were transported on ice to the laboratory, and kept at 1−4 °C prior to being processed for analysis (within 12–24 h).

2.2. General sediment characterisation

The porewater was extracted from the sediment samples by centrifuging at 4000 rpm for 20 min. The pH and E_h of porewater samples was measured within a N_2 filled glove-bag. Additional porewater was passed through a syringe driven 0.45 μm filter and this 'dissolved' fraction was divided into aliquots for analysis. One aliquot was preserved in a sulfide preservation solution (20% zinc acetate in 2 mol/L NaOH) prior to analysis of $\sum S^{2-}$ (which includes H_2S, S^{2-}, HS^- and aqueous sulfide complexes) by the methylene blue method (Cline, 1969). Another aliquot was acidified with HCl to pH <2 for analysis of DOC by combustion on a TOC 5000a instrument (Shimadzu, Kyoto, Japan). A third aliquot was analysed for dissolved PO_4–P by the molybdenum blue method (APHA, 1998).

Sedimentary monosulfides (mainly nanoparticulate-FeS and FeS) were targeted by an acid-volatile sulfide (AVS) extraction, as described by Burton et al. (2007). *Aqua regia* digestion (1:3 HNO_3^-: HCl) was performed on dried, finely ground sediment for total S extraction. Subsamples of dried, ground sediments were subjected to perchloric acid extraction (APHA, 1998) prior to the analysis of total sediment phosphorus (total P) by the molybdenum blue method (Laskov et al., 2007). Total sediment carbon (total C) and total sediment nitrogen (total N) were analysed by sample combustion on an Elementar C, N analyser (Elementar, Vario Macro, Hanau, Germany). Total organic C (TOC) was also measured on the Elementar C, N analyser following the removal of carbonates by 6 mol/L HCl for 24 h (Bisutti et al., 2004). The total inorganic C (TIC) content of the sediment solid phase was calculated by the subtraction of TOC from the total sediment C.

2.3. Laboratory-based deoxygenation experiment

This experiment examined the resuspension of sediment collected from Site 4 (0–20 cm depth interval). Humidified compressed air was bubbled through a polypropylene reaction vessel filled with estuary water until saturated with O_2 (100% O_2). The vessel was placed on a magnetic stir plate for continuously stirring throughout the experiment. The appropriate mass of sediment was added to the estuary water and the logging of O_2 was immediately activated. Dissolved O_2 was continually measured until the suspension reached 0% O_2 saturation. This was replicated 4 times for 10 g/L treatments, 4 times for the 50 g/L treatments, 3 times for the 100 g/L treatments, 2 times for the 200 g/L treatments and once for a control with nil sediment addition. Results are reported as combined averages of the replicate treatments, with the median variability being <2% between replicates. Higher individual variability was observed at <10% O_2 between replicates for the 100 g/L and 200 g/L treatments. The initial rate of oxygen loss was determined by fitting a 2nd order polynomial function to the initial 5 minute (for the 10 and 50 g/L treatments), 2 minute (for 100 g/L treatment) or 1 minute (200 g/L) reaction period and determining the slope of concentration versus time at t = 0 (McKibben et al., 2008).

2.4. Laboratory-based sediment resuspension experiment

This experiment also examined resuspension of homogenised sediment collected from Site 4 (0–20 cm depth interval). Three sediment to water ratios (50 g/L, 100 g/L and 200 g/L) and one control treatment (nil sediment addition) were resuspended in the laboratory with estuary water for 2-weeks. The different sediment to water ratios investigated in the resuspension experiment allowed for a comparison with the field dredge sediment loads. This provides an insight into the water chemistry impacts associated with varying the concentration of certain sediment geochemical components (i.e. AVS). Humidified compressed air was continuously bubbled through the suspensions

throughout the duration of the experiment. Each treatment was run with a replicate.

When monitoring the suspensions, the samples were collected and parameters were measured during the daytime, although the treatments were exposed to diurnal conditions throughout the experiment. Prior to sediment addition (0 h) the alkalinity (HCO_3^-) of the filtered (0.45 μm) porewater and surface water for the resuspensions was measured according to a method by Sarazin et al. (1999). The pH and E_h of the suspensions were also measured prior to sediment addition (0 h), as well as at 1, 2, 3, 4, 24, 48, 72, 96, 168 and 336 h after sediment addition. Homogenised sediment/water suspension samples were collected at these time intervals, along with additional sampling times of 0.33 h and 0.66 h. Each sample was immediately centrifuged (under N_2) at 4000 rpm for 3 min to separate the aqueous and solid phases. The aqueous phase was passed through a 0.45 μm filter (Acrodisc supor membrane) and analysed for PO_4–P by the molybdenum blue method (APHA, 1998) and NH_4^+ by a SAN plus segmented flow autoanalyser (Skalar, Breda, The Netherlands).

The solid phase was subjected to an acid-volatile sulphide (AVS) extraction for determination of the FeS content (Burton et al., 2007). Sulfur speciation at 0, 24 and 72 h was also investigated using S K-edge X-ray absorption near-edge spectroscopy (XANES) at the Taiwan Synchrotron facility (Burton et al., 2009; Prietzel et al., 2011) with full methodology described by Morgan et al. (2012).

2.5. Field dredge event

The South Yunderup approach channel (2.6 km) was dredged and the sediment (~28 000 m^3 in total) disposed of in two separate locations. Here, we investigate the disposal of 70% of the sediment in a submerged area of the estuary between the 30th June and 23rd October 2008. Dredging took place in daylight hours and although scheduled to occur daily, this was dictated by weather and equipment limitations. As such dredging was sometimes halted for several hours, to several days. The sediment was pumped from the estuary floor and transported to the sediment disposal point by large pipes. The water monitoring events from this study only took place when the dredge was in operation. Five water monitoring events were conducted between July 2008 and September 2008 to investigate the impact of the large scale sulfidic sediment disturbance on the water chemistry of the estuary. Site A was located outside of the plume, 100 m up-current from the disposal point, as a reference for undisturbed estuary conditions. Site B was at the point of sediment disposal into the estuary. The remaining sites (C, D, E and F) were sampled approximately every 100 m in a transect across the dredge plume, moving away from the sediment disposal point. The turbidity, E_h, pH, and dissolved O_2 were recorded at each site. Also, the approximate depth of visibility was recorded at each site using a secchi disc with marked depth intervals. Additionally, a water sample was collected from each site. Following filtration (0.45 μm) an aliquot was preserved in 50% (v/v) HNO_3 for the analysis of metals (As, Cd, Co, Cr, Cu, Fe, Mn, Mo, Ni, Pb, Zn) and S by ICP-OES (Perkin–Elmer Optima 7300DV, Connecticut, USA). Detection limits for all metals were <0.003 mg/L while the detection limit for S was 0.3 mg/L. A further aliquot was frozen prior to the analysis of NH_4^+ by a segmented flow autoanalyser and PO_4–P by the molybdenum blue method.

2.6. General methods

All glassware and sampling equipment was soaked in 10% HCl for at least 24 h, and was then rinsed a minimum of three times with Milli-Q water (18.2 MΩ.cm) prior to use. All chemicals used were analytical reagent grade and all reagents and standards were prepared in Milli-Q water. The water content of the homogenised sediment was determined by weight loss over 24 h at 105 °C and sediment properties are reported herein on a dry weight basis. The pH (Cyberscan electrode) and E_h (TPS electrode) were recorded when a steady state was obtained (generally 10 min for E_h and <1 min for pH). The E_h probe was calibrated against Zobell's solution and cleaned regularly with 0.1 mol/L HCl and micro abrasive paper to prevent sulfide contamination of the platinum electrode. E_h readings are reported in mV relative to the standard hydrogen electrode (SHE). Dissolved O_2 was measured with a TPS oxygen probe calibrated in air at 100% O_2.

3. Results

3.1. Sediment properties for the South Yunderup channel

The selected sediment porewater parameters (pH, E_h, $\sum S^{2-}$, NH_4^+, PO_4–P and DOC) are presented in Table 1. The pH was generally neutral to near-neutral, ranging between pH 7 and 8, which is typical of anoxic marine fine-grained sediments (Morse, 1999). The redox conditions of all sediments were reducing (E_h between 100 and −100 mV, SHE) to highly reducing (E_h < −100 mV, SHE) (Table 1). The porewater $\sum S^{2-}$ concentrations were high (median = 3.5 mmol/L; range = 0.1−6.4 mmol/L), as were the porewater DOC concentrations (median = 5.4 mmol/L; range = 0.8–14.9 mmol/L). Porewater NH_4^+ concentrations (median = 6.3 mmol/L, range 0.01–24.4 mmol/L) and PO_4–P concentrations (median = 0.3 mmol/L, range = 0.001–1.5 mmol/L) were also very high compared to most estuarine systems (Kemp et al., 1990; Morin and Morse, 1999).

The sediment AVS, total S, total N, total P, TOC and TIC concentrations are presented in Table 2. In general, the AVS concentrations were elevated (median = 191 μmol/g; range = 34–335 μmol/g) relative to typical estuarine sediments which contain AVS at <100 μmol/g (Burton et al., 2005; Morgan et al., 2012). The total S concentrations ranged between 193 and 677 μmol/g, with a median concentration of 470 μmol/g. The median sediment TIC content ranged from 0.3−2.2 mmol/g across all sites. The TOC in the sediment samples examined here was very high, with a median of 3.6 mmol/g (range = 0.5−6.7 mmol/g). Concentrations were also high for the solid phase N and P, with a median total N concentration of 552 μmol/g (65–1071 μmol/g) and a median total P of 21.5 μmol/g (2.1–36.0 μmol/g).

Table 1
A summary of porewater parameters for sediments collected from the South Yunderup channel, Sites 1–6. The median concentrations for parameters are presented, with the range of concentrations in parentheses.

Site	pH	E_h (mV, SHE)	$\sum S^{2-}$ (mmol/L)	NH_4^+ (mmol/L)	PO_4–P (mmol/L)	DOC (mmol/L)
1	7.3 (7.0−7.6)	−172 (−208, −141)	3.3 (0.3−4.6)	14.9 (2.1−16.4)	0.7 (0.1−1.5)	9.9 (1.9−11.3)
2	7.3 (7.2−7.6)	−160 (−178, 2)	4.0 (0.1−6.4)	6.5 (0.2−14.8)	0.2 (0.0−0.7)	5.5 (0.8−11.3)
3	7.3 (7.0−7.5)	−163 (−315, 71)	5.0 (0.1−6.3)	9.4 (0.5−24.4)	0.4 (0.0−1.1)	5.8 (0.8−14.9)
4	7.5 (7.5−7.7)	−202 (−204, −146)	3.8 (0.4−4.2)	6.2 (0.5−12.6)	0.4 (0.0−0.8)	7.1 (1.0−10.1)
5	7.6 (7.4−7.8)	−196 (−211, −167)	2.3 (1.6−2.9)	1.0 (0.4−1.1)	0.03 (0.02−0.03)	2.8 (2.1−3.1)
6	7.1 (7.0−7.3)	−136 (−153, −118)	[a]	0.02 (0.01−0.03)	0.002 (0.001−0.002)	2.1 (1.6−2.6)

[a] Below detection limit.

Table 2
A summary of solid phase parameters for sediment collected from Sites 1–6 in the South Yunderup channel. The median concentrations for parameters are presented, with the range of concentrations in parentheses.

Site	AVS (μmol/g)	Total S (μmol/g)	Total N (μmol/g)	Total P (μmol/g)	TOC (mmol/g)	TIC (mmol/g)
1	299 (183–331)	527 (486–555)	570 (445–701)	24.6 (23.0–28.7)	3.3 (1.2−4.4)	1.6 (0.9−4.7)
2	147 (34–230)	506 (379–677)	537 (351–744)	20.8 (15.9–24.9)	3.6 (2.6−5.2)	1.4 (0.4−2.7)
3	251 (71–285)	386 (369–463)	740 (635–1071)	28.0 (23.9–36.0)	4.3 (3.5−6.7)	2.2 (1.4−3.4)
4	177 (84–277)	442 (405–509)	594 (521–664)	21.5 (20.4–26.9)	3.7 (3.0−4.7)	1.3 (1.0−2.6)
5	76 (56–104)	442 (236–616)	211 (122–263)	8.1 (6.5–10.2)	1.3 (1.0−1.7)	0.6 (0.2−1.1)
6	87 (39–134)	435 (193–677)	127 (65–189)	4.4 (2.1–6.8)	0.8 (0.5−1.1)	0.3 (0.1−0.6)

3.2. Laboratory-based deoxygenation experiment

The dissolved O_2 concentrations decreased in all treatments following sediment addition (Fig. 2). This rate increased almost linearly as a function of the sediment to water ratio, ranging from 7.9%$_{sat}$ min^{-1} at a sediment to water ratio of 10 g L^{-1} to 203%$_{sat}$ min^{-1} at a corresponding ratio of 200 g L^{-1} (Fig. 2). The average deoxygenation time ranged from was 206 min for the 10 g/L treatments to 2.6 min for the 200 g/L treatments (Fig. 2).

3.3. Laboratory-based sediment resuspension experiment

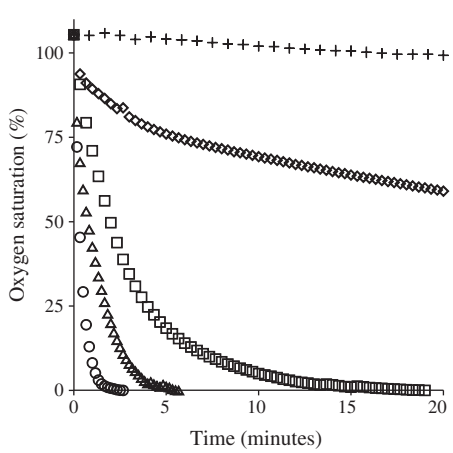
With the addition of the anoxic sediment, the E_h dropped between +50 mV and +150 mV for all treatments (Fig. 3a). Following 1 h of resuspension, the E_h began to steeply increase to reach between +300 mV and +400 mV in all treatments within 24 h. The pH of suspensions remained relatively stable following the sediment addition and only began to decrease after 24 h when the solutions reached more oxidising conditions (Fig. 3b). In treatments with the highest sediment to water ratio (200 g/L) the pH dropped to a minimum of 6.99, before increasing to approximately the same pH (~7.5) that was observed for the end points of the other treatments (Fig. 3b).

The AVS content of the initial homogenised sediment was 145 μmol/g, with 2027 μmol/L of AVS added to the 50 g/L treatments, 3870 μmol/L to the 100 g/L treatments and 7095 μmol/L to the 200 g/L treatments. Following 1 h of resuspension the AVS began to rapidly decrease in all treatments (Fig. 4a). This was consistent with disappearance of FeS and the appearance of more oxidised S species including elemental S and SO_4^{2-}, as measured by XANES spectroscopy (Fig. 4b). AVS was almost completely oxidised in all treatments by 24 h of resuspension, with the majority being oxidised within the first 4 h (Fig. 4).

Table 3 provides a summary for the laboratory resuspension treatments, comparing the AVS addition and H^+ production with the potential buffering capacity provided by porewater HCO_3^-, surface water HCO_3^- and the solid phase TIC. The stoichiometric relationship between FeS consumption and H^+ production demonstrates that the oxidation of 1 mol of FeS is expected to produce 2 mol of H^+ (Eq. (1), Eq. (2)). Considering this, during complete oxidation of AVS, the predicted H^+ production for the 50 g/L treatments is 4.1 mmol/L, for the 100 g/L treatments is 7.7 mmol/L and for the 200 g/L treatments is 14.2 mmol/L (Table 3). A similar concentration of combined porewater and surface water HCO_3^- addition to each treatment was calculated, at 5.2 mmol/L for the 50 g/L treatments, 7.9 mmol/L for 100 g/L treatments and 13 mmol/L for the 200 g/L treatments (Table 3). The solid phase TIC added to each resuspension also provides buffering capacity and can be calculated from the average TIC concentration of the Site 4 sediments. This was approximately 21.0 mmol/L for the 50 g/L treatments, 40.1 mmol/L for the 100 g/L treatments and 73.5 mmol/L for the 200 g/L treatments (Table 3).

Ammonium was rapidly released following the initiation of sediment resuspension (Fig. 5a), with the highest average NH_4^+ concentrations for each treatment being reached at 4 h. Following this, NH_4^+ concentrations remained relatively stable up until 24–100 h, when they began to drop away more rapidly. The highest average NH_4^+ concentration was 0.79 mmol/L for the 50 g/L treatments, 1.43 mmol/L for the 100 g/L treatments and 2.89 mmol/L for the 200 g/L treatments (Fig. 5a). We calculated the maximum NH_4^+ concentration that could be attributed to porewater dilution (plus average blank surface water concentrations) as 0.73 mmol/L for the 50 g/L treatments, 1.36 mmol/L for the 100 g/L treatments and 2.49 mmol/L for the 200 g/L treatments.

Phosphate was also released during the laboratory resuspension (Fig. 5b). Following a peak at 0.33 h, PO_4–P steadily decreased in all



Sediment:water ratio (g/L)	Initial rate of deoxygenation (%/min)	Time to complete deoxygenation (min)
10	7.9	206
50	31	16.5
100	78	4.8
200	203	2.6

Fig. 2. Results from the laboratory-based deoxygenation experiment with 10 g/L (◇), 50 g/L (□), 100 g/L (△) and 200 g/L (○) of wet-weight sediment, with a control (+) of nil sediment addition. Also presented are the initial deoxygenation rates and the time to complete deoxygenation from the experiment. The initial rate of deoxygenation was determined by fitting (all $r^2 > 94$) a 2nd order polynomial to the initial 5 minute (for the 10 and 50 g/L treatments), 2 minute (for 100 g/L treatment) or 1 minute (200 g/L) reaction period.

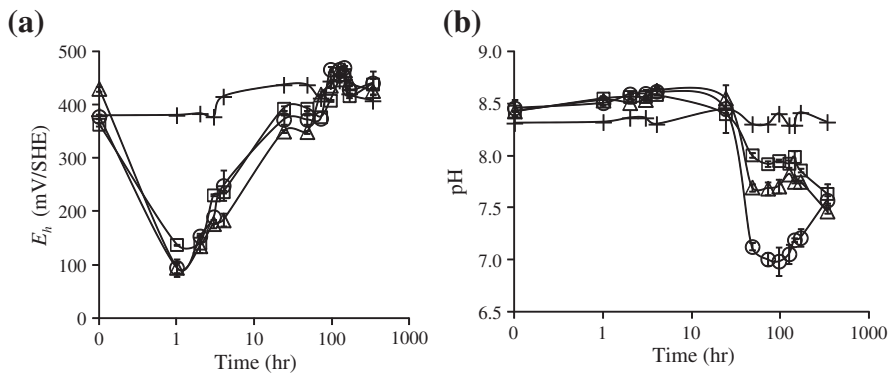


Fig. 3. The (a) redox potential and the (b) pH from the laboratory resuspension experiment in water from the study site. Treatments were 50 g/L (□), 100 g/L (△) and 200 g/L (○) wet-weight sediment, with a control (+) of nil sediment addition. The error bars represent standard deviation between replicates.

treatments, becoming very low to undetectable within 2 h. The highest average PO_4–P was 10.4 μmol/L for the 50 g/L treatments, 14.6 μmol/L for the 100 g/L treatments and 28.5 μmol/L for the 200 g/L treatments (Fig. 5b). However, we calculated an estimated maximum PO_4–P release for each treatment that could be attributed to porewater dilution (plus average blank surface water concentrations) to be higher than this, at 17.7 μmol/L for the 50 g/L treatments, 34.0 μmol/L for the 100 g/L treatments and 61.9 μmol/L for the 200 g/L treatments.

3.4. Field-based dredge monitoring

At the sediment disposal point (Site B) a black plume was consistent with a localised area of increased turbidity (Fig. 6). As such, within 100–200 m from Site B, the turbidity decreased back to the values observed outside of the plume (Fig. 6a). Likewise, an impaired depth of visibility was generally localised to the disposal site, with normal depth of visibility restored within 100–200 m (Fig. 6b). The magnitude of the increased turbidity at the sediment disposal point varied depending on the sampling day, with a maximum increase (up to 473 NTU) observed at the on the 27th of July and a minimum (45.7 NTU) on the 13th August. Based on the relationship measured between nephelometric turbidity units (NTU) and the suspended sediment load in the Brisbane River estuary, sediment to water ratios at the disposal point in this study should range between ~50 mg/L and >200 mg/L (Hossain et al., 2004). This is comparable to the sediment loads used in the laboratory resuspension experiments (see Section 3.3).

The pH and E_h values for the dredge transect during monitoring are displayed in Fig. 7. A localised decrease in redox potential at the point of sediment disposal in the dredge plume was observed (Fig. 7). These lower E_h values observed in the field were consistent with the minimum E_h values observed in the laboratory resuspension experiment (see Section 3.2). The pH was unaffected by distance from disposal site (Fig. 7), with the variation observed between sampling dates being representative of normal estuarine variability.

The release of NH_4^+ and PO_4–P was also observed to be localised at the sediment disposal point in the dredge plume (Fig. 8). There was no detectable release of dissolved trace metals (As, Cd, Co, Cr, Cu, Mn, Mo, Ni, Pb, Zn) during the dredge monitoring. The dissolved iron (Fe) and S concentrations were particularly high at the dredge disposal point on the 13th August compared with other Fe and S values throughout the monitoring period (Fig. 9a). This coincided with the lowest surface water E_h measurement (44 mV) throughout the dredge monitoring. For all other sampling days, dissolved Fe did not vary noticeably across the dredge transect (Fig. 9a). Ignoring

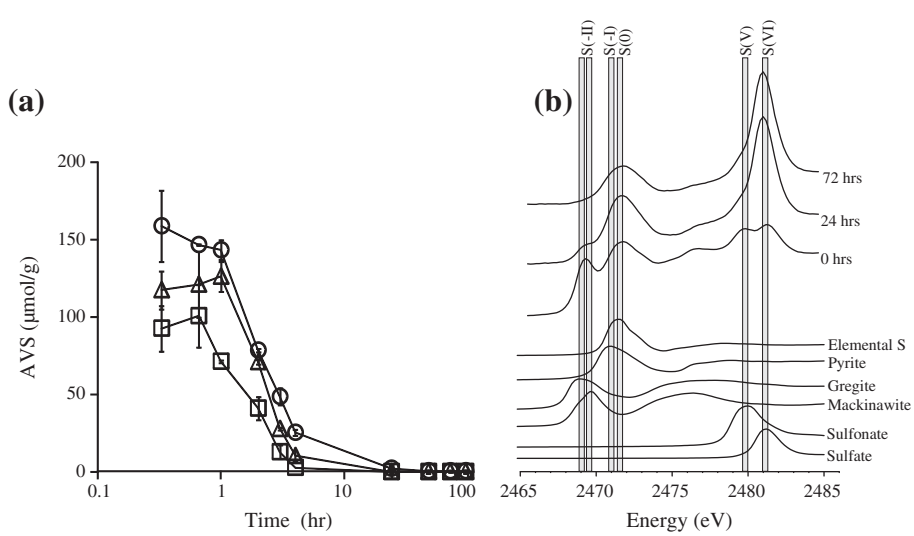


Fig. 4. (a) AVS concentrations during a laboratory resuspension of sediments in site water, with treatments of 50 g/L (□), 100 g/L (△) and 200 g/L (○) wet-weight sediment. The error bars represent standard deviation between replicates. Concentrations of AVS added to each treatment (see Section 2.4) were not graphed to maintain the clarity of the figure. (b) A comparison between the S K-edge XANES peaks for reference phases and samples collected at three time intervals during the laboratory sediment resuspension (100 g/L of wet-weight sediment).

Table 3
A summary of the AVS addition, H^+ production, porewater HCO_3^- addition, surface water HCO_3^- and the solid phase TIC addition for the treatments in the laboratory resuspension.

Treatment (g/L)	AVS addition (mmol/L)	H^+ produced[a] (mmol/L)	Porewater HCO_3^- addition (mmol/L)	Surface water HCO_3^- (mmol/L)	Solid phase TIC addition (mmol/L)
50 g/L	2.0	4.1	2.8	2.4	21.0
100 g/L	3.9	7.7	5.5	2.4	40.1
200 g/L	7.1	14.2	10.6	2.4	73.5

[a] Calculated based on stoichiometric relationship between FeS consumption and H^+ production following complete oxidation of initial AVS.

this single anomalous data point, the dissolved Fe concentration decreased as dissolved S increased ($r^2 = 0.73$, $P < 0.001$) (Fig. 9b).

4. Discussion

4.1. E_h, pH and O_2

It is increasingly recognised that sedimentary S cycling can strongly influence the ecological sustainability of aquatic ecosystems (Åström, 2001; Boman et al., 2008; Brunet, 1996; Burton et al., 2006b, 2006c; Bush et al., 2004; Caraco et al., 1989; Cooper and Morse, 1998; Di Toro et al., 1992; Morgan et al., 2012; Rickard and Morse, 2005; Ward et al., 2004; Macdonald et al., 2004). Several small-scale laboratory-based studies have demonstrated that the resuspension and oxidation of FeS can cause water column deoxygenation and release acidity and trace metals into surface waters (Burton et al., 2006a, 2006b; Saulnier and Mucci, 2000; Simpson et al., 1998; Bush et al., 2004; Fyfe et al., 2007). Despite this, relatively little research exists investigating the influence of large-scale disturbance of FeS-rich sediment on the water chemistry of shallow estuarine systems.

In this study, surface water pH, E_h and dissolved O_2 were investigated during the resuspension of FeS-rich sediment from a eutrophic etuarine system. The rapid consumption of FeS during our laboratory-based sediment resuspension experiment coincided with an increasing E_h following the first hour of sediment resuspension (Figs. 3, 4). This is consistent with previous studies that demonstrate the relatively rapid (<24 h) oxidation of AVS in natural sediments during resuspension events (Burton et al., 2006a; Simpson et al., 1998). While we do not have AVS data for monitoring of the field dredging event, the obvious decrease in E_h at the sediment disposal point (Fig. 7) suggests a localised area of more reducing conditions consistent with the resuspension of the anoxic, FeS-rich sediments. It is possible that the decrease in E_h may have been localised to the sediment disposal point because of the rapid oxidation of AVS, or because of the rapid resettling of sediments (Fig. 6).

A significant negative correlation between Fe and S (Fig. 9) in the field dredge monitoring (apart from one anomalous point) may seem inconsistent with FeS oxidation according to Eq. (1).

$$4FeS_{(s)} + 8O_{2(g)} \rightarrow 4Fe^{2+}{}_{(aq)} + 4SO_4{}^{2-}{}_{(aq)} \quad (1)$$

However, this relationship can be understood by considering that the low dissolved Fe concentrations are likely to be due to the precipitation of Fe (hydr)oxides following the initial release of Fe^{2+} (Eq. 2):

$$Fe^{2+}{}_{(aq)} + 1/4O_{2\,(g)} + 3/2H_2O_{(l)} \rightarrow FeOOH_{(s)} + 2\,H^+{}_{(aq)}. \quad (2)$$

Treatments with a higher FeS addition demonstrated a greater decrease in pH in the laboratory-based sediment resuspension experiment (Fig. 3). This is consistent with previous studies that demonstrate the production of acidity during oxidation of FeS (Burton et al., 2006a). In general, for all sediment to water ratios the laboratory-based resuspension experiment revealed little change in pH as a result of sediment FeS oxidation (Fig. 3). This is due to the buffering capacity provided by both the estuarine water and the sediment porewater and carbonate phases (Table 3). This also explains the near-neutral conditions observed throughout the field dredge monitoring, despite acidity being produced via Eq. (2). From this we can suggest that in an estuarine system, seawater buffering can prevent the severe acidification events that are observed during sulfidic sediment resuspension in freshwater systems (Sullivan et al., 2002). In turn, this prevents the pH-dependent release of metals during the field resuspension of FeS rich sediments (Burton et al., 2006a).

In the laboratory deoxygenation experiment, the decrease in the level of dissolved O_2 saturation was more rapid for treatments with a higher sediment to water ratio (Fig. 2), providing support for previous studies showing water column deoxygenation during oxidation of FeS-rich sediment (Fyfe et al., 2007; Bush et al., 2004). However, we found that the depletion of O_2 from the water column was more rapid (<20 min) than the majority of the AVS oxidation (occurring after ~1 h). This suggests that the initial oxygen demand of the sediments may also be attributed to the rapid oxidation of reduced porewater species such as $\sum S^{2-}$, NH_4^+, methane and reduced humic substances (Johnston et al., 2003; Wong et al., 2010). In contrast to the

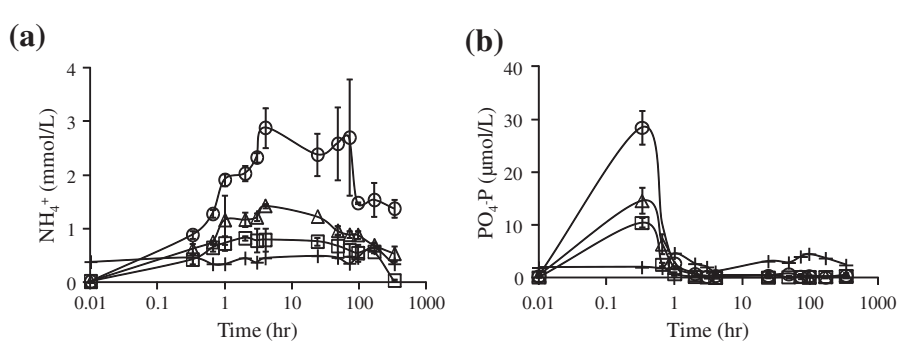


Fig. 5. (a) NH_4^+ and (b) PO_4–P released during a laboratory resuspension of sediments from the South Yunderup channel in water from the study site. Treatments were 50 g/L (□), 100 g/L (△) and 200 g/L (○) of wet-weight sediment, with a control (+) of nil sediment addition. The error bars represent standard deviation between replicates.

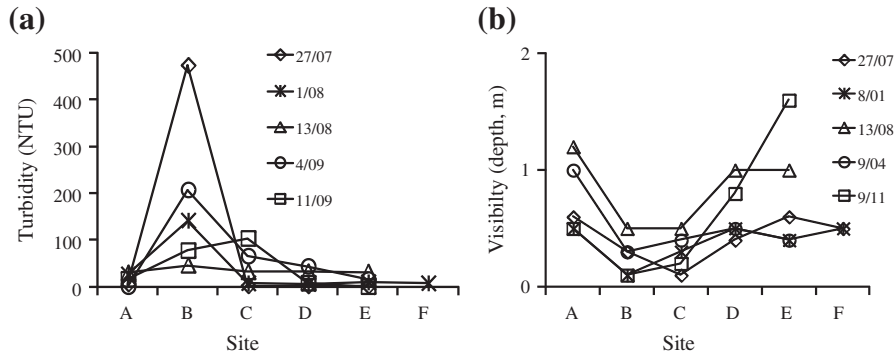


Fig. 6. (a) The turbidity and (b) the visibility for surface water during monitoring of a dredge plume transect. Site A was up current of the sediment disposal point. The sediment disposal point was at Site B, with each site following this being ±100 m from the last. The date of each separate monitoring event is displayed in the dd/mm format.

deoxygenation observed in the closed-system laboratory experiment, the dissolved O_2 in the field dredge monitoring did not vary noticeably. This could be a consequence of re-aeration via water column mixing caused by the wave action observed during sampling.

4.2. Nutrient release

In the organic-rich estuarine sediments examined here, organic matter decomposition is driven largely by SO_4^{2-} reduction (Morgan et al., 2012). Porewater NH_4^+ and PO_4–P are derived from organic nutrient stores that are released during organic C mineralisation (Kristensen and Hansen, 1995). This is further exacerbated by the reduction of Fe (hydr) oxides also releasing PO_4–P, which then often remain dissolved due to a poor affinity for FeS (Jensen et al., 1995; Azzoni et al., 2001; Coelho et al., 2004). Furthermore, the high HS^- concentrations and reducing conditions created during SO_4^{2-} reduction inhibit denitrification pathways and favour the preservation of NH_4^+ in the porewater (Joye and Hollibaugh, 1995; Herbert, 1999). Despite this evident influence of sediment S geochemistry on internal nutrient cycling, nutrient release is not often investigated as a primary environmental issue associated with the resuspension of FeS-rich sediments.

In the current study, the initial rapid increase of NH_4^+ following sediment resuspension can be partially attributed to porewater dilution (Fig. 5). This reflects the extremely high porewater NH_4^+ in the sediments from the South Yunderup channel compared to other similar studies (Table 1). The increase in aqueous NH_4^+ concentrations observed over the initial few hours of sediment resuspension in the laboratory experiment was very similar to the estimated NH_4^+ release expected from porewater dilution. However, as concurrent NH_4^+ loss would also be expected due to chemical and biological oxidation, it is also possible that NH_4^+ may have also been contributed during desorption from the resuspended particles (Morin and Morse, 1999). Ammonium ions strongly adsorb to undisturbed marine sediments, particularly in highly reducing conditions (Morse and Morin, 2005). This has most often been associated with organic matter interactions, which seems likely in the organic-rich sediments examined here (Table 2). The prolonged release of NH_4^+ during laboratory-based sediment resuspension is consistent with a study by Morin and Morse (1999), which demonstrates that a greater proportion of NH_4^+ was released from desorption processes then from porewater dilution effects alone. The decrease in NH_4^+ in the later stages of the experiment may be consistent with chemical oxidation to NO_3^-, or the rapid assimilation of NH_4^+ by biota (Southwell et al., 2010).

In contrast to the laboratory data, NH_4^+ release was localised in the field dredge monitoring and was not observed in the plume +100 m from the sediment disposal point (Fig. 8). Based on turbidity and depth of visibility data (Fig. 6), dredged sediment generally only remained in suspension within 100–200 m of the disposal point. A combination of the high dilution in the well flushed estuary and the transformation of NH_4^+ to nitrate (NO_3^-) in the oxic surface waters may have resulted in the localised release of NH_4^+. Additionally, assimilation of NH_4^+ by biota could result in increased conversion to NO_3^- (Southwell et al., 2010). Further research is needed to quantitatively understand the relative influences of NH_4^+ desorption and conversion to NO_3^- in the field. A smaller scale dredge monitoring programme would also be required to determine how widespread the release of NH_4^+ and NO_3^- is from the dredge disposal point.

PO_4–P was also initially released following sediment addition to laboratory resuspensions (Fig. 5), with higher release corresponding to increased sediment to water ratios. As was suggested for NH_4^+, this initial release of PO_4–P can be attributed to porewater dilution, given the high porewater PO_4–P concentrations in the South

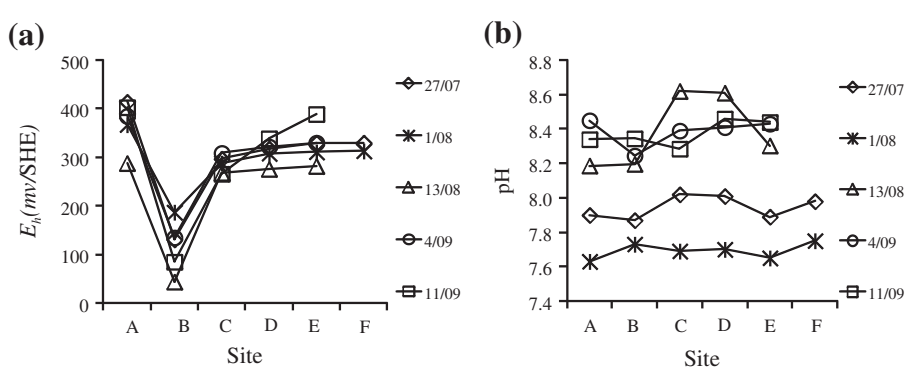


Fig. 7. (a) The E_h and (b) the pH of surface water across a dredge plume transect. Site A was up current of the sediment disposal point. The sediment disposal point was at Site B, with each site following this being ±100 m from the last. The date of each separate monitoring event is displayed in the dd/mm format.

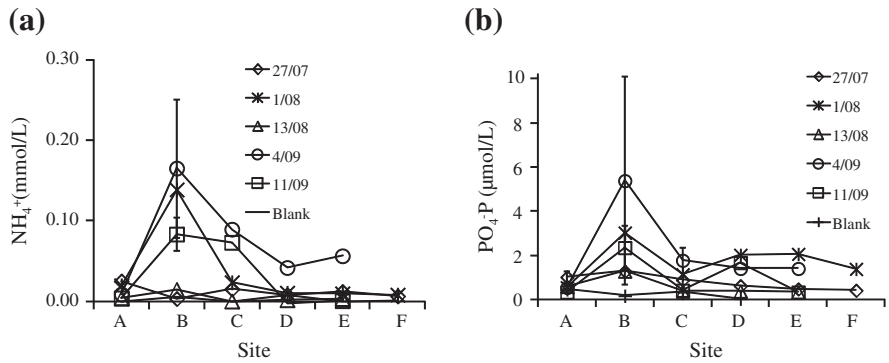


Fig. 8. Concentrations of (a) NH_4^+ and (b) PO_4–P in the surface water across a dredge plume transect. Site A was up current of the sediment disposal point. The sediment disposal point was at Site B, with each site following this being ±100 m from the last. The error bars represent standard deviation between replicate samples. The date of each separate monitoring event is displayed in the dd/mm format.

Yunderup channel sediments (Table 1). It is also possible that some PO_4–P was released by ion exchange from suspended particles (such as organic matter) during oxic resuspension (Sondergaard et al., 1992; Kalnejais et al., 2010). Regardless of the source, PO_4–P was relatively short-lived (<1 h) in solution, with its disappearance coinciding with the onset of AVS oxidation. Furthermore, less PO_4–P was released during the resuspension experiments than would be expected from dilution of the porewater.

It is likely that additional PO_4–P was sequestered by Fe (hydr)oxides that precipitated during FeS oxidation (Eq. 2) (Tengberg et al., 2003). This was supported by our field data, which showed that the highest dissolved PO_4–P concentrations coincided with the lower E_h values at the sediment disposal point. It was here, where the E_h was the lowest, that Fe (hydr)oxides were less likely to be present, resulting in PO_4–P remaining soluble. Dissolved PO_4–P was low to undetectable in the oxic waters away from the disposal point (Fig. 8). This is consistent with the influence of estuary flushing, in addition to the scavenging of PO_4–P by newly-formed Fe (hydr)oxides.

The influence of nutrient release on primary productivity during sediment resuspension is somewhat controversial (Tengberg et al., 2003). It has been suggested that nutrient release during sediment resuspension does not influence long term nutrient concentrations in surface waters (Blackburn, 1997; Sloth et al., 1996). However, others have demonstrated that nutrient released during sediment resuspension has contributed to increased system eutrophication (Morin and Morse, 1999; Cornwell and Owens, 2011; Fanning et al., 1982; Sondergaard et al., 1992). These conflicting views are likely to stem from variations in original sediment characteristics and the system hydrodynamics between studies at the time of dredging (Lohrer and Wetz, 2003). As such, monitoring programmes should be devised depending on the system conditions.

5. Conclusion

There is limited research investigating the influence of FeS-rich sediment resuspension on estuarine water conditions by combined field and laboratory-based investigations. Furthermore, an emphasis is generally placed on surface water deoxygenation and the release of acidity and trace metals during FeS oxidation. In this study, we found that although acidity was produced during AVS oxidation, the buffering capacity of the estuarine water maintained surface waters at a stable, near-neutral pH in the field. This, coupled with the high degree of system flushing, meant that trace metals remained at very low concentrations during the field dredge monitoring programme. Additionally, while water deoxygenation occurred following the resuspension of highly reduced sediments in our laboratory-based experiment; no change in the surface water O_2 was observed during the field-based dredge monitoring.

Despite sedimentary S cycling being coupled with the cycling of N and P (Caraco et al., 1989; Joye and Hollibaugh, 1995; Smolders et al., 2006) nutrient release is rarely investigated as a primary environmental concern during FeS-rich sediment resuspension. Our data demonstrates that, while PO_4–P can be released during sediment resuspension in such systems, the formation of P-scavenging Fe (hydr)oxides by FeS oxidation strongly limits dissolved PO_4–P concentrations. Data from the current study and previous laboratory resuspension experiments indicate that desorption processes may result in NH_4^+ being released from sediments over a longer time period than PO_4–P (Morin and Morse, 1999). However, like PO_4–P, NH_4^+ release was localised to the point of sediment disposal in the field dredge monitoring, indicating that in well-flushed oxygenated estuarine conditions, NH_4^+ release may not be a long-term concern. Further research is needed to investigate the chemical and biological conversion of inorganic N and P to oxidised species,

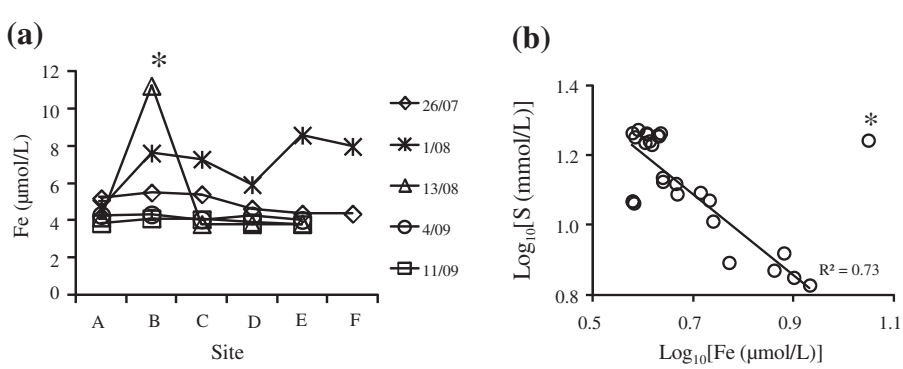


Fig. 9. (a) The dissolved Fe concentrations measured during the field dredge monitoring. (b) The dissolved Fe relative to dissolved S in during the field dredge event. The asterisk (*) on both plots marks an anomalous point that was not included in the regression or test of significance. The date of each separate monitoring event is displayed in the dd/mm format.

as well as the release of organic N and P during large-scale resuspension of the FeS-rich sediments.

This study demonstrates that, alongside the emphasis on acidification, deoxygenation and metal release during the disturbance of FeS-rich sediments, it is important to consider nutrient release, particularly in eutrophic systems. This study also highlights the importance of understanding initial sediment properties and system hydrodynamics when assessing the environmental implications of large-scale sediment disturbance events on water quality.

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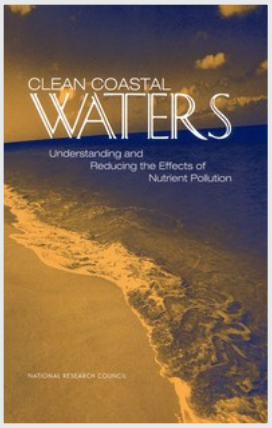
Exhibit G

(Excerpted)



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Committee on the Causes and Management of Coastal Eutrophication
Ocean Studies Board
and
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4

What Are the Effects of Nutrient Over-Enrichment?

KEY POINTS IN CHAPTER 4

This chapter explores the impacts of nutrient over-enrichment and finds:

- The productivity of many coastal marine systems is limited by nutrient availability, and the input of additional nutrients to these systems increased primary productivity.
- In moderation in some systems, nutrient enrichment can have beneficial impacts such as increasing fish production; however, more generally the consequences of nutrient enrichment for coastal marine ecosystems are detrimental. Many of these detrimental consequences are associated with eutrophication.
- The increased productivity from eutrophication increases oxygen consumption in the system and can lead to low-oxygen (hypoxic) or oxygen-free (anoxic) water bodies. This can lead to fish kills as well as more subtle changes in ecological structure and functioning, such as lowered biotic diversity and lowered recruitment of fish populations.
- Eutrophication can also have deleterious consequences on estuaries even when low-oxygen events do not occur. These changes include loss of biotic diversity, and changes in the ecological structure of both planktonic and benthic communities, some of which may be deleterious to fisheries. Seagrass beds and coral reefs are particularly vulnerable to damage from eutrophication and nutrient over-enrichment.
- Harmful algal blooms (HABs) harm fish, shellfish, and marine mammals and pose a direct public health threat to humans. The factors that cause HABs remain poorly known, and some events are entirely natural. However, nutrient over-enrichment of coastal waters leads to blooms of some organisms that are both longer in duration and of more frequent occurrence.
- Although difficult to quantify, the social and economic consequences of nutrient over-enrichment include aesthetic, health, and livelihood impacts.

Nutrient enrichment can have a range of effects on coastal systems. On occasion, in some ecosystems moderate nutrient enrichment can be beneficial because increased primary production can lead to increased fish populations and harvest (Jørgensen and Richardson 1996; Nixon 1998). Far more often, when nutrient enrichment is sufficiently great, the effects are detrimental. In some cases, even small increases in nutrient inputs can be quite damaging to certain types of ecosystems, such as those particularly susceptible to changed conditions (e.g., coral reefs).

Direct and indirect ecological impacts of nutrient enrichment include increased primary productivity, increased phytoplankton biomass, reduction in water clarity, increased incidences of low oxygen events (hypoxia and anoxia), and changes in the trophic structure, trophic interactions, and trophodynamics of phytoplankton, zooplankton, and benthic communities. Harmful algal blooms may become more frequent and extensive. Coral reefs and submerged macrophytic vegetation, such as seagrass beds and kelp beds, may be degraded or destroyed. Fish kills may occur, and more importantly, subtle changes in ecological structure may lead to lowered fishery production. Generally, nutrient over-enrichment leads to ecological changes that decrease the biotic diversity of the ecosystem.

The ecological effects of nutrient over-enrichment can have societal impacts as well, although the economic consequences are generally difficult to quantify. These include aesthetic impacts, such as loss of visually exciting coral reefs and seagrass beds, as well as production of noxious odors and unappealing piles of algal detritus on beaches. Fishery resources can be damaged or lost. Human health is threatened by accumulation of toxins in shellfish. Property can be devalued. This chapter summarizes the societal impacts of nutrient enrichment.

ECOLOGICAL EFFECTS

Increased Primary Productivity

As discussed earlier in this report, eutrophication is a process of increasing organic enrichment of an ecosystem where the increased rate of supply of organic matter causes changes to that system (Nixon 1995). This increased rate of supply is driven by primary productivity. Primary productivity is affected by a variety of factors, including light availability, nutrients, and grazing mortality. The interplay of these factors determines how a coastal marine ecosystem will respond to nutrient additions. (These and other factors that determine an estuary's sensitivity to eutrophication are discussed in detail in Chapter 6.) For many systems, primary productivity is limited largely by nutrient availability, and in these systems

increasing the nutrient input increases the primary productivity rate and often the phytoplankton biomass. As explained in Chapter 3, in the majority of coastal systems—at least in the temperate zone—nitrogen is the element most limiting of primary productivity; consequently, rates of primary production and standing stock of phytoplankton biomass are often directly related to nitrogen inputs.

As noted in a previous report from the National Research Council (NRC 1993a), planners and managers now are often at a disadvantage because "no guidelines exist by which to determine whether coastal marine ecosystems are in fact eutrophic." That report goes on to recommend that coastal eutrophication be judged by some measure of the relationship between phytoplankton biomass (as represented by chlorophyll concentrations) and trophic status, the same approach that is generally used by limnologists for freshwater lakes. Adoption of such an approach would lead to the conclusion that "few estuaries are oligotrophic, many are mesotrophic, and many are extremely eutrophic" (NRC 1993a).

Other authors have suggested similar approaches. For instance, Jaworski (1981) has suggested a lake-based framework of nutrient-loading guidelines that, if met, would tend to keep most estuaries from becoming eutrophic. However, demonstrable harm from human-increased nutrient loading to estuaries has occurred in some systems even when the loadings were low enough not to be called eutrophic by these standards (NRC 1993a).

Nixon (1995) suggested another set of guidelines—these based on measured rates of primary production—for determining whether an estuary is eutrophic. In this classification scheme, estuaries with productivity between 300 and 500 g C m^{-2} yr^{-1} would be considered eutrophic, while those with productivities greater than 500 g C m^{-2} yr^{-1} would be considered hypereutrophic. These guidelines, too, lead to the conclusion that many estuaries are eutrophic or even hypereutrophic.

Increased Oxygen Demand and Hypoxia

Eutrophication is accompanied by an increased demand for oxygen. Some of this increased oxygen demand is due to the greater respiration of the increased biomass of plants and animals that are supported in the nutrient-loaded ecosystem. Much of it is often due to respiration of bacteria (in both the water column and sediments) that consume the organic matter produced by the greater plant production. If the loss of oxygen caused by increased respiration is not offset by the direct introduction of additional oxygen by photosynthesis or mixing processes, then hypoxia or anoxia occurs. Biologists generally refer to the situation where some oxygen is present but where dissolved oxygen levels are less than or

equal to 2.0 milligrams per liter (mg l^{-1}) as hypoxia. Anoxia is the complete absence of oxygen.

Hypoxia and anoxia are more likely to occur in summer because warming of the water column can lead to stratification and the formation of a barrier that prevents the introduction and mixing of oxygen from surface waters. Also, the solubility of oxygen decreases and oxygen demand (respiration rate) generally increases as temperature increases.

As noted earlier, many studies of the biological impacts of reduced dissolved oxygen concentrations have used 2.0 mg l^{-1} as the cut off for designating conditions as hypoxic (e.g., Pihl et al. 1991, 1992; Schaffner et al. 1992), because below this threshold there are severe declines in the diversity and abundance of species in the systems. There is evidence, however, that 2.0 mg l^{-1} may not be a universal threshold. For example, the results of a study of biological resources in Long Island Sound, New York, revealed that 3.0 mg l^{-1} was the threshold level for finfish and squid (Howell and Simpson 1994). A study of the benthic community in Corpus Christi Bay, Texas, indicated that dissolved oxygen concentrations less than 3.0 mg l^{-1} should be the operational definition of hypoxia in that system (Ritter and Montagna 1999), and that a single value of dissolved oxygen as a water quality standard for estuarine waters may not be appropriate.

Many states have standards for dissolved oxygen levels in aquatic systems that are well above the limits used to define hypoxia. The Florida Department of Environmental Protection mandates that the average level of dissolved oxygen that must be maintained in marine waters designated for the commercial harvest of shellfish, recreation, and for the maintenance of healthy fish and wildlife is greater than 5.0 mg l^{-1} in a 24-hour period and never less than 4.0 mg l^{-1}. Although this level may seem conservative, in the absence of detailed information for a system it may be appropriate.

The occurrence of hypoxic and anoxic bottom waters, particularly in the coastal zone, has become a major concern in recent years because it appears that the frequency, duration, and spatial coverage of such conditions have been increasing, and this increase is thought to be related to human activities (Diaz and Rosenberg 1995). Zones of reduced oxygen can disrupt the migratory patterns of benthic and demersal species, lead to reduced growth and recruitment of species, and cause large kills of commercially important invertebrates and fish (NRC 1993a). Such conditions can also lead to an overall reduction in water quality, thereby affecting other coastal zone activities such as swimming and boating. Reports of a "dead zone," an extensive area of reduced oxygen levels covering an expanse originally of some 9,500 km^2 in the Gulf of Mexico (Rabalais et al. 1991), have focused attention on the problem of coastal zone hypoxia. By

the summer of 1999, the hypoxic area in the Gulf of Mexico had grown to an area of 20,000 km^2 (Rabalais personal communication).

Researchers studying the Chesapeake Bay have said since the 1980s that the occurrence of hypoxic and anoxic bottom waters has increased in association with nutrient inputs (Taft et al. 1980; Officer et al. 1984). More recent studies examined pollen distribution, diatom diversity, and the concentration of organic carbon, nitrogen, sulfur, and acid-soluble iron in sediment cores from the mesohaline portion of the bay (Cooper and Brush 1991). The cores represented a 2,000-year history of the bay. Changes in the concentration of organic components and pollen abundance coincided with the new settlement by Europeans in the late 1700s. This period was marked by major land clearing in the watershed, which likely promoted increases rates of sedimentation, mineralization, and nitrification, and an increase in agricultural activity and the use of manures. Analysis of the sediment cores indicated a shift in the phytoplankton community from centric to pennate diatoms for this time period, and this was interpreted as evidence of increased nutrient input to the bay. This historical perspective indicates a role for nutrients in the occurrence of hypoxia in Chesapeake Bay. As discussed in Chapter 5, the input of nutrients to Chesapeake Bay has probably accelerated even more in the last several decades due to increased use of inorganic fertilizer and increased combustion of fossil fuels and the resulting atmospheric deposition of nitrogen.

The northern Adriatic Sea and northern Gulf of Mexico are two other coastal systems that have experienced increasing episodes of hypoxia (Justic et al. 1993; Turner and Rabalais 1994). Both systems are affected by river flow, the Po River in the case of the former and the Mississippi River in the latter. In both systems researchers have documented a seasonal increase in primary productivity in surface waters that was related to nutrients and river flow; this increase was followed by hypoxia in the bottom waters. The hypoxia onset, however, lagged peak river flow in the Gulf of Mexico and Adriatic Sea by two and four months, respectively. This difference in the lag period was ascribed to greater depth of the water column in the Adriatic Sea and differences in the downward flux of organic matter. Again, the evidence showed that the introduction of new nutrients in the river flow contributed to the development of hypoxia in these systems, but stratification of the water column was a necessary condition.

There also is evidence that increased nutrient loading has contributed to the occurrence of hypoxia in Florida Bay and the Florida Keys (Lapointe at al. 1990; Lapointe and Clark 1992). The most severe cases of hypoxia were found in the canal and seagrass systems closest to the discharge areas. Increased nitrogen levels were associated with increased growth of nutrient-limited phytoplankton, whereas high levels of soluble reactive P

were associated with increased growth of macroalgae and tropical seagrasses. Lapointe and Matzie (1996) showed that episodic rainfall events led to higher submarine discharge rates that were followed within days by hypoxic oxygen levels.

Shifts in Community Structure Caused by Anoxia and Hypoxia

The occurrence of hypoxic and anoxic bottom waters may also lead to shifts in benthic and pelagic community structure due to the mortality of less mobile or more sensitive taxa, reduction of suitable habitat, and shifts in predator-prey interactions (Diaz and Rosenberg 1995). Hypoxia plays a major role in the structuring of benthic communities because species differ in the sensitivity to oxygen reduction (Diaz and Rosenberg 1995). The response of species to reduced oxygen availability also depends on the frequency and duration of these events. With short bouts of hypoxia, some large or very motile species are able to adjust to or move away from the stress.

Hypoxia tends to shift the benthic community from being dominated by large long-lived species to being dominated by smaller opportunistic short-lived species (Pearson and Rosenberg 1978). In addition, recurring hypoxia may limit successional development to colonizing communities. In such systems more organic matter is available for remineralization by the microbial community. This can decrease the amount of energy available for benthic recruitment when hypoxia and anoxia disappears. Zooplankton that normally vertically migrate into bottom waters during the day may be more susceptible to fish predation if they are forced to restrict their activity to the oxic surficial waters. Roman et al. (1993) concluded that the vertical distribution of copepods in the Chesapeake Bay was altered by the presence of hypoxic bottom waters. Moreover, an hypoxic or anoxic bottom layer may constitute a barrier that de-couples the life cycle of pelagic species (e.g., diatoms, dinoflagellates, and copepods) that have benthic resting stages (Marcus and Boero 1998).

In a controlled eutrophication experiment (Doering et al. 1989), the structure of the zooplankton community was affected by the presence or absence of an intact benthic community. In the absence of an intact benthic community, holoplanktonic forms, especially higher level predators, dominated, whereas meroplanktonic forms were more evident in the presence of an intact benthic community. Although the data did not identify the mechanism behind these shifts, the differences likely reflected alterations in the coupling of the benthic and pelagic environments (nutrient as well as life cycle linkages) (Marcus and Boero 1998).

Changes in predator-prey interactions in the water column can also lead to shifts in energy flow. Increased fish predation on zooplankton can

release grazing pressure on the phytoplankton and increase the deposition of organic matter to the sediments. If the duration and severity of the hypoxia is not sufficient to cause mortality of the macrobenthos, the increased supply of organic matter to the benthic system could fuel the growth of benthic fauna and demersal fish populations at the expense of pelagic fisheries. On the other hand, extended hypoxic and anoxic events could lead to the demise of the macrobenthos and the flourishing of bacterial mats. The loss of burrowing benthic organisms that irrigate the sediments and the presence of an extensive bacterial community may alter geochemical cycling and energy flow between the benthic and pelagic systems (Diaz and Rosenberg 1995). For example, the flux of nitrogen out of the sediments is affected by the rates of nitrification and denitrification, and these processes depend on the naturally oxic and anoxic character of the sediments.

Changes in Plankton Community Structure Caused Directly by Nutrient Enrichment

Nutrient over-enrichment can also change ecological structure through mechanisms other than anoxia and hypoxia. Phytoplankton species have wide differences in their requirements for and tolerances of major nutrients and trace elements. Some species are well adapted to low-nutrient conditions where inorganic compounds predominate, whereas others thrive only when major nutrient concentrations are elevated or when organic sources of nitrogen and phosphorus are present. Uptake capabilities of major nutrients differ by an order of magnitude or more, allowing the phytoplankton community to maintain production across a broad range of nutrient regimes. A decrease in silica availability in an estuary and the trapping of silica in upstream eutrophic freshwater ecosystems can occur as a result of eutrophication and thus nitrogen and phosphorus over-enrichment. This decrease in silica often limits the growth of diatoms or causes a shift from heavily silicified to less silicified diatoms (Rabalais et al. 1996). Given these changes in the cycling of nitrogen, phosphorus, and silica, it is no surprise that the phytoplankton community composition is altered by nutrient enrichment (Jørgensen and Richardson 1996).

The consequences of changes in phytoplankton species composition on grazers and predators can be great, but in general these are poorly studied. As noted by Jørgensen and Richardson (1996):

> Any eutrophication induced change in the species composition of the phytoplankton community which leads to a change in size structure of the phytoplankton community will potentially affect energy flow in the entire ecosystem. Thus, eutrophication can, at least in theory, play an

> important role in dictating whether the higher trophic levels in a given system are dominated by marketable fish or by jellyfish. . . . Little is actually known about the effects of eutrophication on the size structure of the phytoplankton community under various conditions, but this is an area that warrants further research.

In particular, a change from diatoms toward flagellates, which may tend to result during eutrophication as the silica supply is diminished, may be deleterious to food webs supporting marketable forms of finfish (Greve and Parsons 1977). On the other hand, if the silica supply remains high enough, moderate eutrophicaton can encourage more growth by diatoms and lead to higher fish production (Doering et al. 1989; Hansson and Rudstam 1990).

Looking beyond the major nutrients, it is also evident that phytoplankton species have variable requirements for nutritional trace elements or have different tolerances for toxic metals (Sunda 1989), and the effects of these elements can be affected by dissolved organic matter (DOM) concentrations. One example of this effect is seen with copper, which is highly toxic to marine organisms and is often significantly elevated in harbors and estuaries due to anthropogenic inputs. Copper is strongly bound to organic chelators in seawater, and this lowers copper's biological availability and consequently its toxicity (Sunda 1989). Moffett et al. (1997) studied copper speciation and cyanobacterial distribution and abundance in four harbors subject to varying degrees of copper contamination from anthropogenic sources. Cell densities of cyanobacteria, one of the most copper-sensitive groups of phytoplankton, declined drastically in high copper waters compared to adjacent unpolluted waters. Because of the variability in the concentrations of the natural organic ligands that bind the copper, relatively small changes in the total copper concentration (7 to 10 times) among the study sites were associated with much larger (greater than 1000 times) changes in the free Cu^{2+} activity, the biologically available form.

The bioavailability of metals such as iron also can be affected by human activities, including nutrient pollution and the resulting eutrophication, and this in turn can affect phytoplankton species composition and thus ecosystem structure and function. Iron is an essential element for algae, and is required for electron transport, oxygen metabolism, nitrogen assimilation, and DNA, RNA, or chlorophyll synthesis. Organic ligands are needed to keep iron in solution at the pH of seawater, as iron hydroxides have extremely low solubility and tend to transform into stable crystalline forms that do not directly support algal growth. DOM plays a critical role in enhancing the bioavailability of iron in seawater. A variety of factors can affect DOM levels in estuaries and coastal systems, but in general

eutrophication results in higher DOM levels—due to higher levels of primary production, leakage of DOM from phytoplankton, release as phytoplankton are eaten or decompose—with concomitant changes in iron availability.

Although the potential impacts of nutrient enrichment on phytoplankton community structure in the field seem obvious, there are few well-documented examples. This is because the changes in community structure often are gradual and easily obscured by fluctuations in other controlling factors, such as temperature, light, or physical forcings.

Long-term data sets offer another insight into possible changes since they allow sustained trends to be detected in spite of short-term variability caused by weather or other environmental forcings. For example, a 23-year time series off the German coast documented the general enrichment of coastal waters with nitrogen and phosphorus, as well as a four-fold increase in the nitrogen:silicon and phosphorus:silicon ratios (Radach et al. 1990). This was accompanied by a striking change in the composition of the phytoplankton community, as diatoms decreased and flagellates increased more than ten-fold. Other data from nearby regions showed a change in the phytoplankton species composition accompanying a shift in the nitrogen:phosphorus supply ratio along the Dutch coast (Cadée 1990), as well as increased incidence of summer blooms of the marine haptophyte *Phaeocystis* after a shift from phosphorus-limitation to nitrogen-limitation (Riegman et al. 1992). Nutrient status, particularly phosphorus-limitation, is now believed to be a major factor driving colony formation in this genus. Experiments performed with cultures of *Phaeocystis* demonstrate that free-living solitary cells outcompete the more harmful colonial forms in ammonium- and phosphate-limited conditions, whereas colonies dominate in nitrate-replete cultures. This suggests that free-living *Phaeocystis* cells would be prevalent in environments that are regulated by regenerated nitrogen, whereas colonial forms would require a nitrate supply and thus would be associated with "new" nitrogen such as that supplied by pollution.

Another long-term perspective on nutrient enrichment on phytoplankton community structure is seen in recent data examining the abundance of dinoflagellate cysts in bottom sediments of Oslofjord, Norway (Dale et al. 1999). Dinoflagellate cysts are an important group of microfossils used extensively for studying the biostratigraphy and paleoecology of sediments. In this study, dinoflagellate cyst records were analyzed from sediment cores that covered a period of anthropogenic nutrient enrichment that began in the mid- to late-1800s, was heaviest from 1900 to the 1970s, and then diminished from the mid-1970s to the present. Over the period of nutrient and organic enrichment, cyst abundance in the sediments doubled and a marked increase in one species in particular,

Lingulodinium machaerophorum (=*Gonyaulax polyedra*), from less than 5 percent to around 50 percent of the assemblage was noted. In the core considered most representative of general water quality in the inner fjord (Figure 4-1), these trends reversed back to pre-industrial levels during the 1980s and 1990s when improved sewage treatment took effect. Other changes in the phytoplankton community no doubt occurred that were not revealed with this approach, but the cyst record nevertheless demonstrates substantial changes in the abundance and composition of a major phytoplankton class.

Although at times changes in community structure are directly the result of nutrient enrichment, sometimes they are an indirect result of other changes caused by increased nutrients. For instance, a change in the phytoplankton community in the form of selection for different species can be caused directly by increased nitrogen. On the other hand, a change in phytoplankton community structure can be caused indirectly by increased nitrogen, because higher levels of nitrogen increase productivity, which increases dissolved organic carbon, which in turn causes changes in the community structure. Generally it is difficult to determine whether community structure changes are direct or indirect.

Harmful Algal Blooms

Among the thousands of species of microscopic algae at the base of the marine food web are a few dozen that produce potent toxins or that cause harm to humans, fisheries resources, and coastal ecosystems. These species make their presence known in many ways, ranging from massive blooms of cells that discolor the water, to dilute, inconspicuous concentrations of cells noticed only because of the harm caused by their highly potent toxins. The impacts of these phenomena include mass mortalities of wild and farmed fish and shellfish, human intoxications or even death from contaminated shellfish or fish, alterations of marine trophic structure through adverse effects on larvae and other life history stages of commercial fisheries species, and death of marine mammals, seabirds, and other animals.

"Blooms" of these algae are sometimes called red tides, but are more correctly called HABs, and are characterized by the proliferation and occasional dominance of particular species of toxic or harmful algae. As with most phytoplankton blooms, this proliferation results from a combination of physical, chemical, and biological mechanisms and interactions that are, for the most part, poorly understood. HABs have one feature in common, however; they cause harm either due to their production of toxins or to the way the cells' physical structure or accumulated biomass affect co-occurring organisms and alter food web dynamics. This descrip-

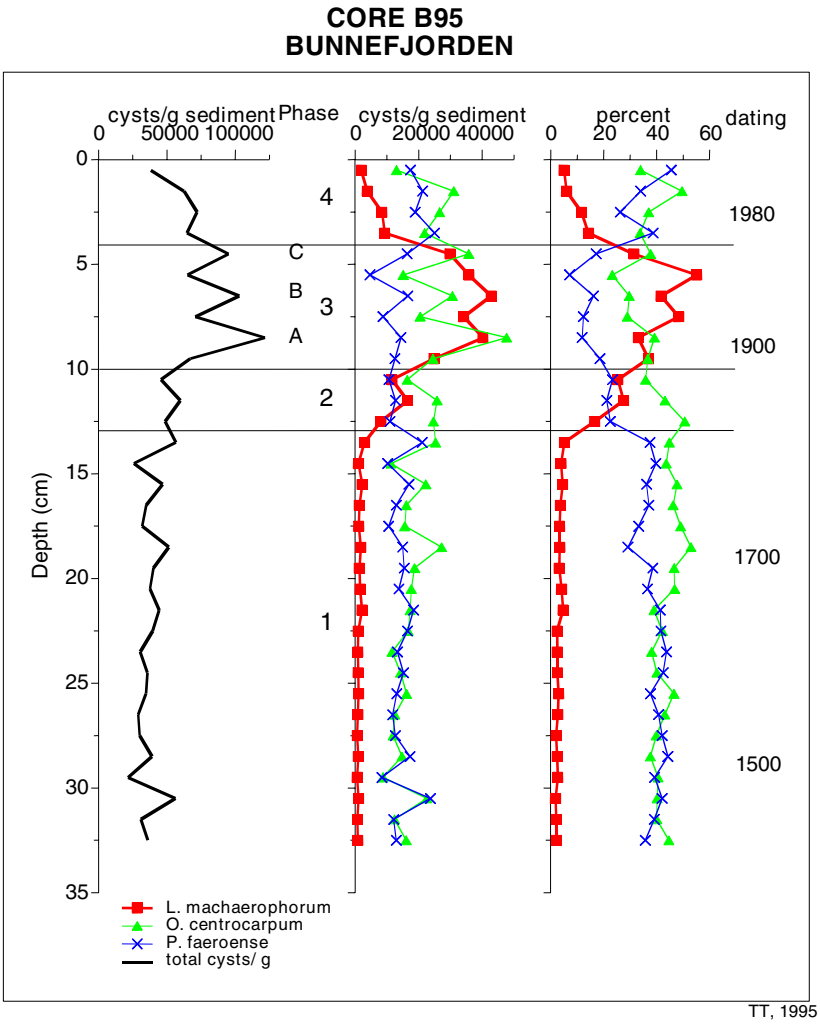


FIGURE 4-1 Water quality in Oslofjord, Norway, as indicated by the changes in the proportions of dinoflagellate cysts in sediment cores. The left panel shows the concentrations of all cysts combined, the middle panel shows the concentration of the three most important species (*Lingulodinium machaerophorum*, *Operculodinium centrocarpum*, and *Peridinium faeroense*), and the right panel shows the relative abundance of those species in percent. Eutrophication of Oslofjord resulted in a doubling of total cyst abundance and by a marked increase in *L. machaerophorum* from less than five percent to nearly 50 percent of the assemblage. These trends reversed with improved water quality in the 1980s and 1990s (modified from Dale et al. 1999).

tor applies not only to microscopic algae but also to benthic or planktonic macroalgae that can proliferate and cause major ecological impacts, such as the displacement of indigenous species, habitat alteration, and oxygen depletion. The causes and effects of macroalgal blooms are similar in many ways to those associated with harmful microscopic phytoplankton species.

HAB phenomena take a variety of forms. One major category of impact occurs when toxic phytoplankton are filtered from the water as food by shellfish that then accumulate the algal toxins to levels that can be lethal to humans or other consumers. These poisoning syndromes have been given the names paralytic, diarrhetic, neurotoxic, and amnesic shellfish poisoning (PSP, DSP, NSP, and ASP). A National Research Council report (NRC 1999b) summarized the myriad human health problems associated with toxic dinoflagellates. In addition to gastrointestinal and neurological problems associated with the ingestion of contaminated seafood, respiratory and other problems may arise from toxins that are released directly into seawater or become incorporated in sea spray. Whales, porpoises, seabirds, and other animals can be victims as well, receiving toxins through the food web from contaminated zooplankton or fish.

Another type of HAB impact occurs when marine fauna are killed by algal species that release toxins and other compounds into the water or that kill without toxins by physically damaging gills. Farmed fish mortalities from HABs have increased considerably in recent years, and are now a major concern to fish farmers and their insurance companies. The list of finfish, shellfish, and wildlife affected by algal toxins is long and diverse (Anderson 1995) and accentuates the magnitude and complexity of the HAB phenomena. In some ways, however, this list does not adequately document the scale of toxic HAB impacts, as adverse effects on viability, growth, fecundity, and recruitment can occur within different trophic levels, either through toxin transmitted directly from the algae to the affected organism or indirectly through food web transfer. This is because algal toxins can move through ecosystems in a manner analogous to the flow of carbon or energy.

Yet another HAB impact is associated with blooms that are of sufficient density to cause dissolved oxygen levels to decrease to harmful levels as large quantities of algal biomass fall to the sediment and decay as the bloom declines. Oxygen levels can also drop to dangerous levels in "healthy" blooms due to algal respiration at night. Estuaries and nearshore waters are particularly vulnerable to low dissolved oxygen problems during warm summer months, especially in areas with restricted flushing.

One of the explanations given for the increased incidence of HAB outbreaks worldwide over the last several decades is that these events are

a reflection of pollution and eutrophication in estuarine and coastal waters (Smayda 1990). Some experts argue that this is evidence of a fundamental change in the phytoplankton species composition of coastal marine ecosystems due to the changes in nutrient supply ratios from human activities (Smayda 1990). This is clearly true in certain areas of the world where pollution has increased dramatically. It is perhaps real, but less evident, in areas where coastal pollution is more gradual and unobtrusive. A frequently cited dataset from an area where pollution is a significant factor is from Tolo Harbor in Hong Kong, where population growth in the watershed grew six-fold between 1976 and 1986. During that time, the number of observed red tides increased eight-fold (Lam and Ho 1989). The underlying mechanism is presumed to be increased nutrient loading from pollution that accompanied human population growth. A similar pattern emerged from a long-term study of the Inland Sea of Japan (Box 4-1).

Both the Hong Kong and Island Sea of Japan examples have been criticized, since both could be biased by changes in the numbers of observers through time, and both are tabulations of water discolorations from algal blooms, rather than just toxic or harmful episodes. Nevertheless, the data demonstrate that coastal waters receiving industrial, agricultural, and domestic effluents, which frequently are high in plant nutrients, do in fact experience a general increase in algal growth.

Nutrients can stimulate or enhance the impact of toxic or harmful species in several ways. At the simplest level, toxic phytoplankton may increase in abundance due to nutrient enrichment but remain as the same relative fraction of the total phytoplankton biomass (i.e., all phytoplankton species are stimulated equally by the enrichment). In this case, we would see an increase in HAB incidence, but it would coincide with a general increase in algal biomass. Alternatively, some contend that there has been a *selective* stimulation of HAB species by nutrient pollution. This view is based on the nutrient ratio hypothesis (Smayda 1990), which argues that environmental selection of phytoplankton species has occurred because human activities have altered nutrient supply ratios in ways that favor harmful forms. For example, diatoms, the vast majority of which are harmless, require silicon in their cell walls, whereas most other phytoplankton do not. As discussed in Chapter 3, silica availability is generally decreased by eutrophication. In response to nutrient enrichment with nitrogen and phosphorus, the nitrogen:silicon or phosphorus:silicon ratios in coastal waters have increased over the last several decades.

Diatom growth in these waters will cease when silicon supplies are depleted, but other phytoplankton classes (which have more toxic species) can continue to proliferate using the "excess" nitrogen and phosphorus. The massive blooms of *Phaeocystis* that have occurred with increasing frequency along the coast of western Europe are an example of this phe-

BOX 4-1
Red Tides and Eutrophication in the Inland Sea of Japan

A prominent example of the link between eutrophication and increased HABs is seen in a long-term data set of red tides in the Inland Sea of Japan. As Japanese industrial production grew rapidly in the late 1960s and early 1970s, pollution of coastal waters also increased. Currently, the number of visible red tides increased from 44 per year in 1960 to more than 300 a decade later, matching the pattern of increased nutrient loading from pollution (Figure 4-2). Japanese authorities instituted effluent controls in the early 1970s through the Seto Inland Sea Law, resulting in a 70 percent reduction in the number of red tides that has persisted to this day (Okaichi 1997).

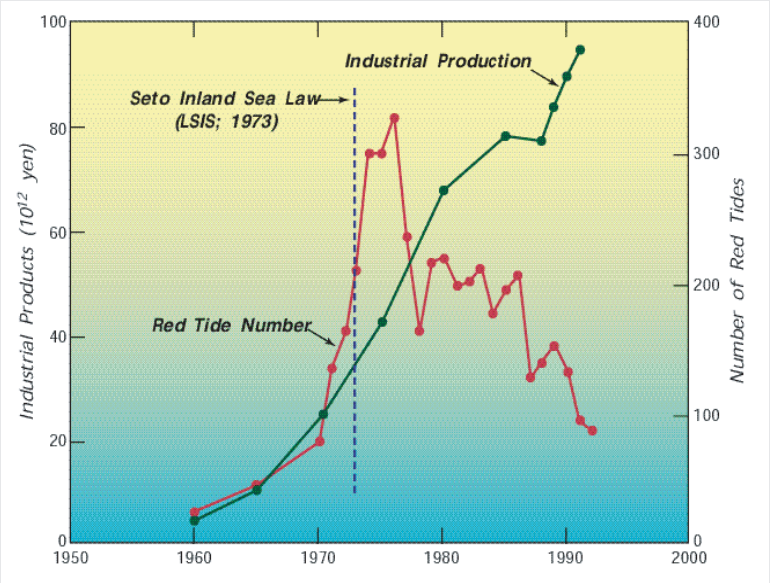


FIGURE 4-2 Changes in the number of visible red tides in the Inland Sea of Japan, 1960-1990 (Okaichi 1997; used with permission from Terra Scientific Publishing).

nomenon (Lancelot et al. 1998). Other examples include the fish killing blooms of *Chattonella* species, which have caused millions of dollars of damage in the Seto Inland Sea, though the frequency and severity of these outbreaks has decreased since pollution loading was reduced and eutrophication abated somewhat (Okaichi 1997).

Another frequently cited example of the potential linkage between HABs and pollution involves the recently discovered "phantom" dinoflagellate *Pfiesteria*. In North Carolina estuaries and in the Chesapeake Bay, this organism has been linked to massive fish kills and to a variety of human health effects, including severe learning and memory problems (Burkholder and Glasgow 1997). A strong argument is being made that nutrient pollution is a major stimulant to outbreaks of *Pfiesteria* or *Pfiesteria*-like organisms because the organism and associated fish kills have occurred in watersheds that are heavily polluted by hog and chicken farms and by municipal sewage. The mechanism for the stimulation appears to be two-fold. First, *Pfiesteria* is able to take up and use some of the dissolved organic nutrients in waste directly (Burkholder and Glasgow 1997). Second, this adaptable organism can consume algae that have grown more abundant from nutrient over-enrichment. Even though the link between *Pfiesteria* outbreaks and nutrient pollution has not been fully proven, the evidence is strong enough that legislation is already in various stages of development and adoption to restrict the operations of hog and chicken farms in order to reduce nutrient loadings in adjacent watersheds. *Pfiesteria* has thus provided the justification needed by some agencies to address serious and long-standing pollution discharges by nonpoint sources, which heretofore have avoided regulation.

Degradation of Seagrass and Algal Beds and Formation of Nuisance Algal Mats

Many coastal waters are shallow enough that benthic plant communities can contribute significantly to autotrophic production if sufficient light penetrates the water column to the seafloor. In areas of low nutrient inputs, dense populations of seagrasses and perennial macroalgae (including kelp beds) can attain rates of net primary production that are as high as the most productive terrestrial ecosystems (Charpy-Roubaud and Sournia 1990). These perennial macrophytes are less dependent on water column nutrient levels than phytoplankton and ephemeral macroalgae, and light availability is usually the most important factor controlling their growth (Sand-Jensen and Borum 1991; Dennison et al. 1993; Duarte 1995). As a result, nutrient enrichment rarely stimulates these macrophyte populations, but instead causes a shift to phytoplankton or bloom-forming benthic macroalgae as the main autotrophs. Fast-growing micro- and

macroalgae with rapid nutrient uptake potentials can replace seagrasses as the dominant primary producers in enriched systems (Duarte 1995; Hein et al. 1995). The biotic diversity of the community generally decreases with these nutrient-induced changes (Figure 4-3A&B).

Over the last several decades, nuisance blooms of macroalgae ("seaweeds") in association with nutrient enrichment have been increasing along many of the world's coastlines (Lapointe and O'Connell 1989). Phytoplankton biomass and total suspended particles increase in nutrient-enriched waters and reduce light penetration through the water column to benthic plant communities. Epiphytic microalgae become more abundant on seagrass leaves in eutrophic waters and contribute to light attenuation at the leaf surface, as well as to reduced gas and nutrient exchange (Tomasko and Lapointe 1991; Short et al. 1995; Sand-Jensen 1977). Ephemeral benthic macroalgae have light requirements that are significantly less than either seagrasses or perennial macroalgae, and also can shade perennial macrophytes such as seagrasses and contribute to their decline (Markager and Sand-Jensen 1990; Duarte 1995). These nuisance algae are typically filamentous (sheet-like) forms (e.g., *Ulva*, *Cladophora*, *Chaetomorpha*) that can accumulate in extensive thick mats over the seagrass or sediment surface, and this can lead to destruction of these submerged aquatic seagrass systems. Massive and persistent macroalgal blooms ultimately displace seagrasses and perennial macroalgae through shading effects (Valiela et al. 1997). The nuisance algae also wash up on beaches, creating foul-smelling piles.

In addition to shading, seagrass distribution in eutrophic waters is influenced by increased sediment sulfide concentrations resulting from decomposition in anoxic organic-rich sediments. Elevated sediment sulfide has been shown experimentally to reduce both light-limited and light-saturated photosynthesis, as well as to increase the minimum light requirements for survival (Goodman et al. 1995). Both effects interact with increased light attenuation to decrease the depth penetration of seagrasses in eutrophic waters.

Decreased photosynthetic oxygen production at all light levels also reduces the potential for oxygen translocation and release to the rhizosphere, and creates a positive feedback that reduces sulfide oxidation around the roots, further elevating sediment sulfide levels. In Florida, chronic sediment hypoxia and high sediment sulfide concentrations have been associated with the decline of the tropical seagrass *Thalassia testudinum* (Robblee et al. 1991). Sulfide also may reduce growth and production of seagrasses by decreasing nutrient uptake and plant energy status, as has been shown for salt marsh grasses (Bradley and Morris 1990, Koch et al. 1990).

Declines in seagrass distribution caused by decreased light penetra-

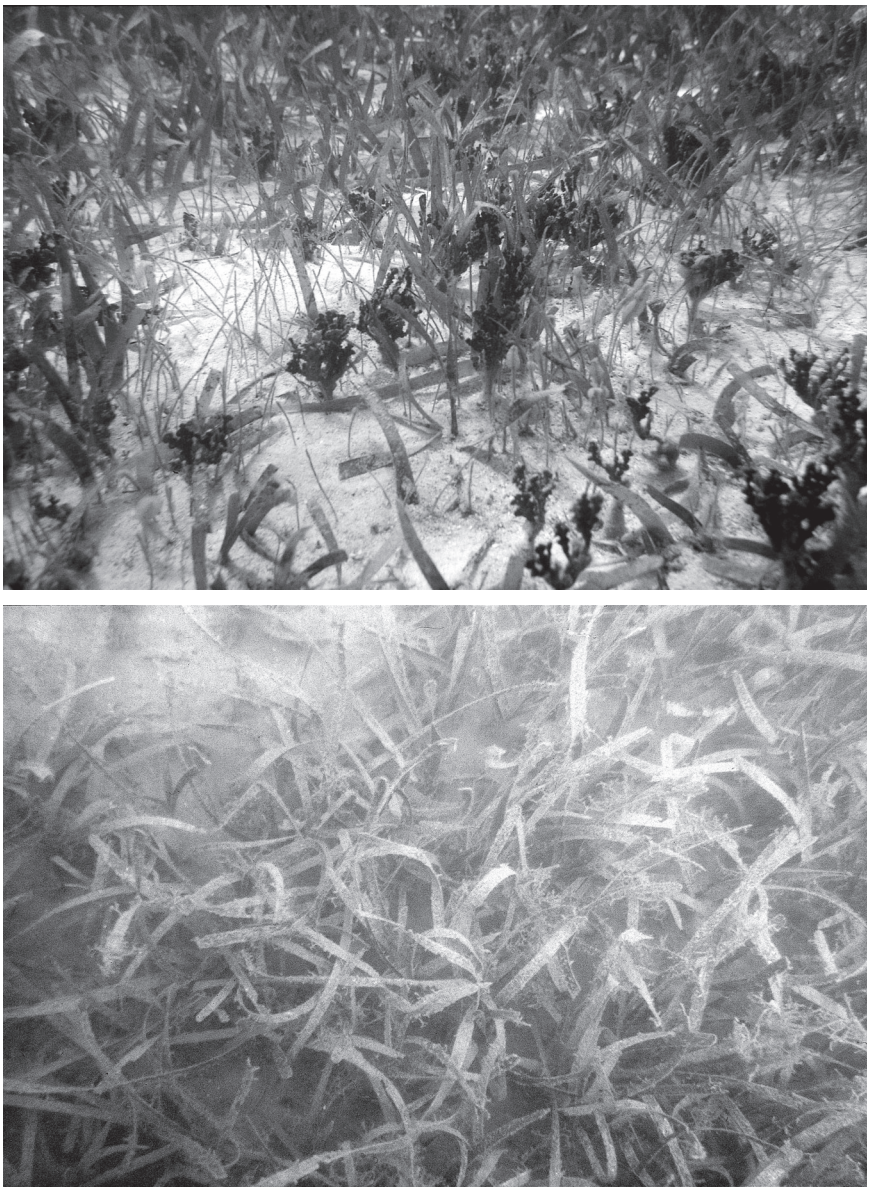


FIGURE 4-3 (A) The bottom-dwelling plants of a marine ecosystem that received natural rates of nitrogen addition. Note the high diversity of these plants and their spacing. (B) The bottom-dwelling plants of a marine ecosystem that received high rates of nitrogen input. Note that there are few plant species, and that the leaves of these are covered with a thick layer of algae (photos by R. Howarth; Vitousek et al. 1997).

tion in deeper waters or changes in community composition prompted by the proliferation of benthic macroalgae in shallower waters will have significant trophic consequences. Seagrass roots and rhizomes stabilize sediments, and their dense leaf canopy promotes sedimentation of fine particles from the water column. Loss of seagrass coverage increases sediment resuspension and causes an efflux of nutrients from the sediment to the overlying water that can promote algal blooms. Seagrasses also provide food and shelter for a rich and diverse fauna, and reduced seagrass depth distribution or replacement by macroalgal blooms will result in marked changes in the associated fauna (Thayer et al. 1975; Norko and Bonsdorff 1996).

In addition, where mass accumulations of macroalgae occur, their characteristic bloom and die-off cycles influence oxygen dynamics in the entire ecosystem. As a result, eutrophic shallow estuaries and lagoons often experience frequent episodic oxygen depletion throughout the water column rather than the seasonal bottom-water anoxia that occurs in stratified, deeper estuaries (Sfriso et al. 1992; D'Avanzo and Kremer 1994). Benthic macroalgae also uncouple sediment mineralization from water column production by intercepting nutrient fluxes at the sediment-water interface (Thybo-Christesen et al. 1993; McGlathery et al. 1997) and can outcompete phytoplankton for nutrients (Fong et al. 1993). Except during seasonal macroalgal die-off events in these shallow systems, phytoplankton production is typically nutrient-limited and water column chlorophyll concentrations are uncharacteristically low despite high nutrient loading (Sfriso et al. 1992).

Coral Reef Destruction

Coral reefs are among the most productive and diverse ecosystems in the world. They grow as a thin veneer of living coral tissue on the outside of the hermatypic (reef-forming) coral skeleton. The world's major coral reef ecosystems are found in nutrient-poor surface waters in the tropics and subtropics. Early references to coral reef ecosystems preferring or "thriving" in areas of upwelling or other nutrient sources have since been shown to be incorrect (Hubbard, D. 1997). Rather, high nutrient levels generally are detrimental to "reef health" (Kinsey and Davies 1979) and lead to phase shifts away from corals and coralline algae toward dominance by algal turf or macroalgae (Lapointe 1999). For example, some offshore bank reefs in the Florida Keys that contained more than 70 percent coral cover in the 1970s (Dustan 1977) now have only about 18 percent coral cover; turf and macroalgae now dominate these reefs, accounting for 48 to 84 percent cover (Chiappone and Sullivan 1997). Reduced

Exhibit H

NORTH CAROLINA
Environmental Quality NC DEQ

Algal Blooms

Report an algae bloom on your phone,tablet or PC

(https://survey123.arcgis.com/share/c23ba14c74bb47f3a8aa895f1d976f0d?portalUrl=https://ncdenr.maps.arcgis.com)

OR

Contact regional office staff to report an algae bloom (http://deq.nc.gov/contact/regional-offices)

Algae are responsive to the physical and chemical conditions in the aquatic environment. Sometimes their rapid reproduction causes nuisance growths or blooms. Most blooms occur when favorable environmental conditions exist, such as an extended photoperiod during summer, sufficient nutrients, and slow-moving or stagnant waters.

Algal blooms have dramatic effects on water chemistry, most notably pH and dissolved oxygen (DO). When algae remove carbon dioxide during photosynthesis they raise the pH by increasing the level of hydroxide. The opposite reaction occurs during respiration when carbon dioxide is produced lowering hydroxide and lowering the pH. Therefore, high pH (greater than 8.0) can be an indicator of photosynthesis by large quantities of algae.

Algal blooms produce large amounts of oxygen during photosynthesis that may lead to supersaturated levels of DO in the water column. Conversely, during respiration, algal blooms remove the DO from the water column, which may lead to little or no oxygen in

the water column. These conditions can also be created when a large number of algae die and decompose. Supersaturation of DO (greater than 110% saturation) can also be an indicator of photosynthesis by large quantities of algae, particularly during the mid-to-late afternoon.

Algae are a concern in drinking water supplies and reservoirs. Some algae, such as Microcystis, produce toxins and have been linked with the deaths of livestock and domestic pets*. They can also cause taste and odor problems, water discoloration, or form large mats that can interfere with boating, swimming, and fishing. Algae and their blooms may be associated with fish kills.

*See the Dept. of Health and Human Services' website (https://epi.dph.ncdhhs.gov/oee/a_z/algae.html) for more information regarding bluegreen algae. Questions concerning possible human health effects should be directed to Kennedy Holt (mailto:Kennedy.Holt@dhhs.nc.gov), Harmful Algal Blooms Coordinator, N.C. Dept. of Health and Human Services. (http://www.ncdhhs.gov/)

Helpful links related to algal blooms:

Frequently Asked Questions Regarding Cyanobacteria (Bluegreen algae)/Cyanotoxins (https://files.nc.gov/ncdeq/Water%20Quality/Water_Sciences/Frequently-Asked-Questions-Cyanobacteria.pdf)

Identifying Cyanobacterial (Bluegreen) Algal Blooms (https://files.nc.gov/ncdeq/Water%20Quality/Water_Sciences/Identifying-Cyanobacterial-Blooms.pdf)

Precautions around blue-green algal blooms (http://epi.publichealth.nc.gov/oee/a_z/algae.html)

Precautions around fish kills (http://epi.publichealth.nc.gov/oee/a_z/fishkills.html)

Latest news release related to algal blooms in the Albemarle Sound area:

July 1, 2020: State Officials Warn of Potential Algal Bloom in Chowan County (https://deq.nc.gov/news/press-releases/2020/07/01/state-officials-warn-potential-algal-bloom-chowan-county)

Ecosystems Branch

Algae & Aquatic Plants (/about/divisions/water-resources/water-resources-data/water-sciences-home-page/ecosystems-branch/algae-aquatic-plants)

Algal Blooms (/about/divisions/water-resources/water-resources-data/water-sciences-home-page/ecosystems-branch/algal-blooms)

- Algal Bloom Map (/about/divisions/water-resources/water-resources-data/water-sciences-home-page/ecosystems-branch/algae-aquatic-plants/algal-bloom-events)

Ambient Monitoring System (/about/divisions/water-resources/water-resources-data/water-sciences-home-page/ecosystems-branch/ambient-monitoring-system)

AMS Quality Assurance Project Plan (/about/divisions/water-resources/water-resources-data/water-sciences-home-page/ecosystems-branch/ams-quality-assurance-project-plan)

Monitoring Coalition Program (/about/divisions/water-resources/water-resources-data/water-sciences-home-page/ecosystems-branch/monitoring-coalition-program)

Random Ambient Monitoring System (/about/divisions/water-resources/water-resources-data/water-sciences-home-page/ecosystems-branch/random-ambient-monitoring-system)

Wetland Information & Projects (/about/divisions/water-resources/water-resources-science-and-data/water-sciences-home-page/ecosystems)

Fighting Algal Blooms

After an absence of 25-30 years, algal blooms have returned to parts of the Chowan River, Edenton Bay, Albemarle Sound, Little River, Perquimans River, and Pasquotank River. The summer blooms in 2015-2019 triggered state advisories for swimming and consuming fish. We are working to determine the specific causes of the blooms, and to provide timely and practical information for blooms in 2020.

Current News

View Division of Water Resources July 9, 2020 algal bloom report **for the Chowan River.**

View Regional Water Quality Data
Citizen scientists have been collecting water quality data at sites across the region since 2018 in order to help determine nutrient hot spots that may be contributing to algal blooms. Click here to see the map and data.

Algal Blooms in Albemarle Waters -- What We Know and What We Need to Study
The ARC&D, Albemarle Commission, and Chowan Soil and Water hosted a meeting in Edenton on February 5th, 2020 to share current research and information on algal blooms in area waters. Thirty three scientists, conservation professionals and local government representatives participated in the meeting, which provided clear direction for future research to identify the causes of, and solutions to, the algal blooms. Click here to read a 2-page fact sheet.

Algal blooms are fed by warm temperatures, sunlight and too much nutrients in the water, mainly nitrogen and phosphorus carried by stormwater runoff. Stormwater in our region flows through drainage canals, creeks and rivers to the Albemarle and Pamlico Sounds. Stormwater can be exposed to nutrients and pesticides from agricultural fields and lawns, oil and grease from roads, parking lots, and other pollutants from septic tank systems, solid waste storage and processing sites, and commercial properties.

Widespread and persistent algal blooms negatively impact recreational boating, fishing and nature tourism, which are important drivers for regional economic growth. Annual, widespread and persistent algal blooms also may lower waterfront property values. Lake Champlain in Vermont is a current, well-documented example.

A 2016 study sponsored by the Albemarle-Pamlico National Estuary Partnership (APNEP) provides information on the economic value of natural resources in the region. The value of many natural resources in the region would be negatively impacted by annual, widespread and persistent algal blooms. Read the report: Economic Valuation of the Albemarle-Pamlico Watershed's Natural Resources.

Each resident can help reduce pollutants carried in stormwater. Please read more about what you can do to reduce pollutants and help prevent algal blooms in our shared waters.

(ans Paerl and Dr.
ences: Quantitative

Evaluation of Nutrient Sources to the Albemarle Sound System. The study was funded by the Clean Water Management Trust Fund under a regional planning grant to the Albemarle Commission.

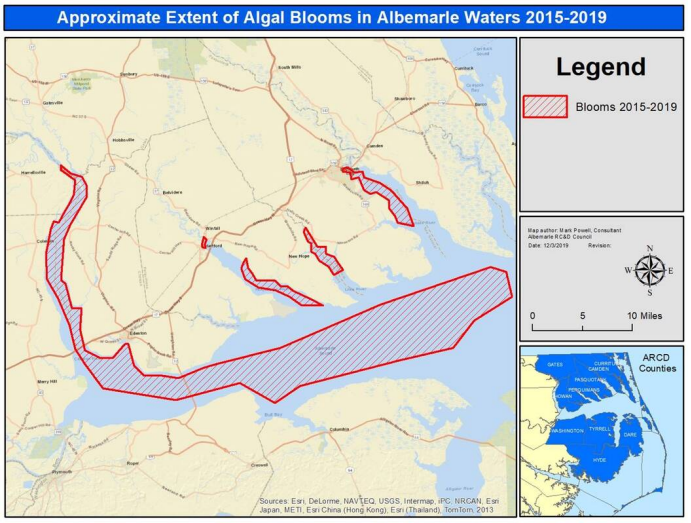
Algal Blooms -- What Have We Learned?
Thirty-three scientists, conservation professionals and local government officials came together in Edenton February 5th, 2020 to share current research and knowledge about algal blooms in Albemarle waters. This was the third meeting in a 3-year planning grant funded by the Clean Water Management Trust Fund. Click here to read the meeting notes.

Two-Page Fact Sheet on Algal Blooms
Learn more about algal blooms.

Good Water Quality Starts at Home!
The algal blooms in our waterways are a result of poor water quality. The blooms are fed by warm temperatures, and too much nutrients in the water, mainly nitrogen and phosphorus. The nutrients are often carried by stormwater and may come from a number of sources including residential yards, agricultural operations, and commercial and industrial sites. Identifying the specific sources of the nutrients, and the actions to reduce them, requires good science and broad public participation. However, each resident can take specific actions **now** to help improve water quality.
Read the Fact Sheet on What You Can Do to Love Your River

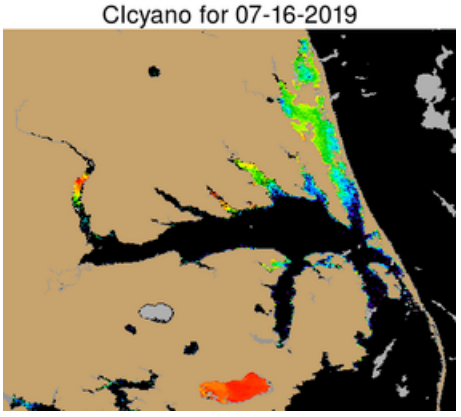
Water Quality Call To Action
Eight counties around the Albemarle Sound and the Albemarle District of Soil and Water, which covers five counties, have adopted and sent to legislators a resolution to strengthen critical drainage and water quality infrastructure. Now individuals can sign the resolution on-line. With algal blooms popping up across the region, please take a minute to make your voice heard:
https://www.greensavesgreen.org/call-to-action

Help Map Algal Blooms!
Click here to help report and map algal blooms in Albemarle waters at the NC Water Resources Division website.



Report a fish kill or algal bloom to DEQ Water Resources

National Centers for Coastal Ocean Science
Harmful Algal Bloom Monitoring System
Cyanobacterial Monitoring in the Albemarle Sound.



Partnership to Monitor Water Quality in Albemarle Waters

The ARC&D is collaborating with the Albemarle Commission, Chowan-Edenton Environmental Group (CEEG), Green Saves Green, Perquimans County Waterway Watch, Soil and Water Conservation Districts, and local governments to monitor water quality in rivers and creeks in the region. Citizen scientists are collecting water samples from key locations and sending them to two labs for analysis. Identifying nutrient hotspots and the sources of nutrients are key steps for developing an effective program to combat algal blooms. Click here to see the map and data.

Algal Blooms in Albemarle Waters -- What We Know and What We Need to Study
The ARC&D, Albemarle Commission, and Chowan Soil and Water hosted a meeting in Edenton on February 5th, 2020 to share current research and information on algal blooms in area waters. Thirty three scientists, conservation professionals and local government representatives participated in the meeting, which provided clear direction for future research to identify the causes of, and solutions to, the algal blooms. Click here to read a 2-page fact sheet.

Algal Blooms -- What Have We Learned?
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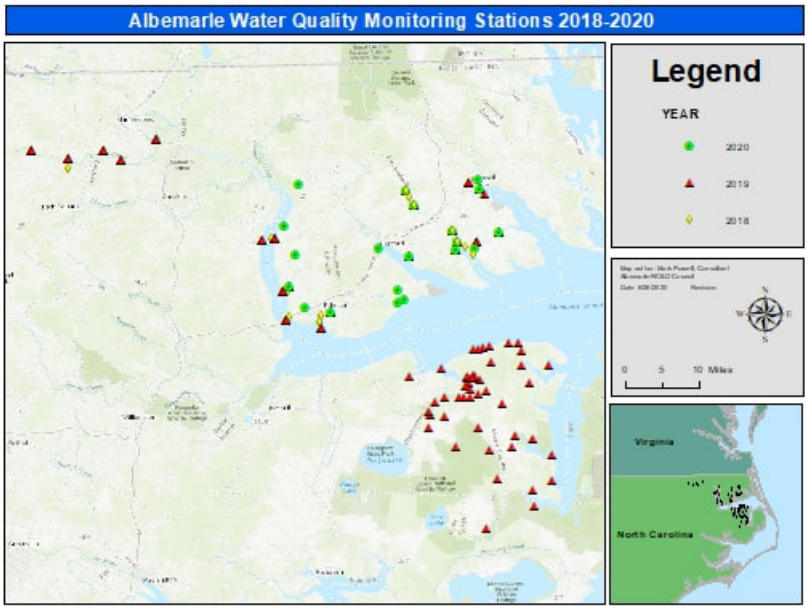
read the meeting notes.

Algal Bloom Planning Meeting - January 30th, 2019 in Edenton

Eighteen people from universities, state agencies, non-profit organizations, and volunteer groups gathered in Edenton on January 30th, 2019 to share information on the potential causes of, and solutions to, algal blooms in Albemarle waters, which have occurred each year since 2015. Research is focusing on nutrient hotspots in the Chowan and Pasquotank River Basins.

At the CEEG event on 8/25/18, Mark Powell, ARC&D Consultant provided information on the broad partnership of local and regional non-profits, state and federal agencies, Soil and Water Conservation Districts, towns, counties, and local farmers that are working together to determine the drivers of algal blooms, and to implement Best Management Practices to reduce nutrients and sediment flowing to creeks, rivers, bays and sounds. The current planning effort is funded through a grant from the Clean Water Management Trust Fund and matching funds from the Albemarle Commission. Click here to see the PowerPoint presentation.

This regional effort is funded by grants from the Clean Water Management Trust Fund and the US Fish and Wildlife Service - Partners for Fish and Wildlife, and the many volunteer citizen scientists who are donating their time.



Algal blooms were a major problem in the 1970's and 1980's.

"... we need the support of every citizen who lives in the Chowan River Basin to make the Chowan Restoration Project successful. The nutrients we must control don't come from one or two specific sources, and all sources must do their share in the clean up. Restoration may mean changes in some of our farming practices, a change in some municipal sewage treatment plants, and changes in industrial waste disposal practices. In the short run, it may seem inconvenient to take these actions. In the long run, it will benefit every citizen in the Chowan River Basin." **Governor Jim Hunt, 1979.** Read the 1979 NC and VA Chowan River Restoration Plan

Algal Blooms in Albemarle Waters - What We Know and Do Not Know

To understand why the blooms have returned, NCDEQ Water Resources is analyzing water quality data from monitoring stations on the Chowan River, Little River and Albemarle Sound. The preliminary analysis of data is helping us better understand what is changing in Albemarle waters, how the changes may be contributing to algal blooms, and the action items needed to effectively address the blooms.

Read a summary: Algal Blooms in Albemarle Waters - What We Know and Do Not Know

What are algal blooms?

Algae are very tiny, often microscopic, plants or plantlike organisms that live in water or damp areas. One type of freshwater algae increasingly seen in North Carolina is cyanobacteria (blue-green algae), which may actually look reddish-brown as well as bright green or blue-green. Many types of algae [illegible]sh in water bodies with poor water flow, especially during the hot months of the year.

The NC Department of Health and Human Services (DHHS) reports that there are no documented reports of people getting sick from blue-green algae in North Carolina. However, DHHS recommends that people follow these common-sense, practical precautions around blue-green algae blooms:

- Don't wade or swim in water containing visible blooms, and avoid direct contact with dense mats of algae.
- Don't drink untreated water or let children, livestock or pets get into or drink untreated water.
- Make sure children are supervised at all times when they are near water. Drowning, not exposure to algae, remains the greatest hazard of water recreation.
- If you do come into contact with the algae or water around a bloom, simply rinse off with fresh water as soon as possible.

DHHS has guidelines specifically for the protection of children and dogs.

The Centers for Disease Control has extensive information on harmful algal blooms including fact sheets and posters.

- Fact Sheet: Cyanobacteria Blooms FAQs
- Fact Sheet: Animal Safety Alert: Cyanobacteria Blooms.

See the the US EPA site Monitoring and Responding to Cyanobacteria and Cyanotoxin in Recreational Waters for more information.

What causes algal blooms?

Algal blooms need:

- Sunlight
- Hot Temperatures
- Slow-moving water
- Nutrients (nitrogen and phosphorus)

Nutrient pollution from human activities makes the problem worse, leading to more severe blooms that occur more often.

What can I do?

Visit the US EPA website on nutrient pollution to learn how you can help prevent algal blooms:

- In your home
- In your yard
- In your community
- In your classroom

Become a **citizen scientist** by helping the Chowan-Edenton Environmental Group monitor and report algal blooms.
Email: ceeg2007@gmail.com

Report a fish kill or algal bloom to DEQ Water Resources

NC Department of Environmental Quality (DEQ) has an outreach and education toolkit for stormwater and runoff pollution.
NCDEQ Water - Education and Technical Assistance Portal

- Water U Know! - What is algae? How does a wastewater plant work? the science of water and lots more ...
- Ground Water - Dig deep - great things are brewing below the surface
- It's Our Water - Water quality curriculum for North Carolina 8th grade science and high school Earth/Environmental Science
- Project Wet - Water Education for Teachers
- Stream Watch - This program encourages neighbors, civic groups and businesses to adopt a local stream
- Improving Water Quality in Your Own Back Yard
- Water Quality - We All Play a Part
- Caring for Your Lawn and the Environment (NC State / NC Cooperative Extension link)

NCDEQ Stormwater & Runoff Pollution Citizen Resources
NCDEQ Algal Bloom Webpage
NCDEQ Fact Sheets

- NC DEQ Algae & Aquatic Plants
- NC DEQ Algae & Aquatic Plant Fact Sheets
- NC DEQ Stormwater Brochure

NC Sea Grant water quality fact sheets:

- Water Quality: Development & Stormwater Fact Sheet
- Water Quality: Estuaries & Polluted Runoff Fact Sheet
- Stormwater Ponds: Improving Aesthetics, Value & Function Fact Sheet
- Trees & Plants: Benefits to the Community

Albemarle-Pamlico National Estuary Partnership

- Albemarle-Pamlico National Estuary Partnership Comprehensive Conservation and Management Plan 2012 - 2022
- Economic Analysis of the Costs and Benefits of Restoration and Enhancement of Shellfish Habitat and Oyster Propagation in North Carolina

The Importance of Constructed Wetlands for Filtering Stormwater

The Albemarle RC&D Council partners with local governments, state and federal agencies, schools and other non-profit groups in 10 counties around the Albemarle and Pamlico Sounds to implement projects that balance resource conservation and economic development. In response to increasing development in the region and its impact on water quality, the RC&D Council and its partners have secured millions of dollars of grant funds to protect land and water resources. Projects have restored eroded shorelines, conserved wetland forests, developed watershed management plans, and constructed about 70 acres of stormwater wetlands on commercial, residential and public properties. The wetlands also serve as outdoor environmental education classrooms for local schools, county and town planners, and the general public. For more information, please click this link to read about some of the land and water conservation projects that the council and its partners have implemented over the last 45 years.

The Albemarle RC&D Council is working with partners in Perquimans and Pasquotank counties to improve water quality in the Little River watershed. The project constructed 6,800 ft of in-stream wetlands on main drainage canals in 2016, and 3,200 ft in-stream wetland will be constructed in 2018 with a new EPA 319 grant. Read more about the effort to restore the Little River watershed.

The Importance of Riparian Forests for Storing and Filtering Stormwater

Riparian forests, commonly cypress and gum in NE NC, are critical for storing and filtering stormwater, and providing key habitat for fish and wildlife. Riparian forests are slow growing and there is a lack of information on how recent, wide-spread clearcutting is impacting water temperature and nutrient release into waterways, in particular the Chowan River.

Key research questions related to the algal blooms include:

- Are there new sources of agricultural, residential, and industrial discharges in the Chowan watershed that are contributing significantly to the algal blooms?
- Is the clearcutting of riparian forests in the Chowan watershed significantly increasing water temperature in the Chowan River and its tributaries?
- Is increased decomposition of soils in riparian clearcuts causing a release of nitrogen and phosphorus in quantities that are stimulating algal blooms?

A swamp forest harvest study (Ensign and Mallin, 2001) helps answer some of these important questions: Stream Water Quality Changes Following Timber Harvest in a Coastal Plain Swamp Forest. Important results include:

Compared with the control creek, the post-clearcut Goshen Swamp displayed significantly higher suspended solids, total nitrogen, total phosphorus, total Kjeldahl nitrogen and fecal coliform bacteria, and significantly lower dissolved oxygen over a 15 month period. Longer-term deleterious effects included recurrent nuisance algal blooms that had not been present during the 2 1/2 years before the clearcut. Although a 10m uncut buffer zone was left streamside, this was insufficient to prevent the above impacts to stream water quality.

A 1983 study of land-use, nutrient yield and eutrophication in the Chowan River basin found that swamp forests removed 83% of the total N and 51% of the total P in streams passing through these wetlands (Craig and Kuenzler, 1983). The study concluded that due to the importance of swamp forests as nutrient buffers, special protection should be given to these areas. Read the 1983 study by Craig and Kuenzler.

In a 2014 study, Sweeney and Newbold found that based on the literature on eight major stream or streamside ecosystem factors (properties, components, or functions), that streamside forest buffers ≥30 m wide are needed to protect water quality, habitat, and biotic features of streams associated with watersheds ≤100 km2 , or about fifth order or smaller in size. Read the 2014 study.

For more information on the importance of riparian forest buffers read: North Carolina's RIPARIAN BUFFERS: A Scientific Review
The 2016 report offers these key findings:

- Buffer width is crucial: riparian buffers with widths of 100 ft to 165 ft have been found to reduce total nitrogen loadings to streams by as much as 85% or more.
- Pollutant removal efficiencies decrease sharply as buffer width decreases. North Carolina research in the Neuse basin coastal plain found that 49 ft buffers reduced nitrogen by 48%, while 26 ft buffers only reduced nitrogen by 28%.
- For 50 ft buffers such as those currently required by state rules, North Carolina research indicates that the factors most directly shaping their effectiveness are whether they have a high water table and extensive woody vegetation.

Bloom Reports in 2017-2019

Algal Bloom Update 9/11/19 Albemarle Sound. Read the report from NC DEQ.

Algal Bloom Update 9/12/19 Chowan River. Read the report from NC DEQ.

Algal Bloom Update 7/5/19 - Little River. Read the report from NC DEQ.

Algal Bloom Update 6/20/19 - Chowan River Near US 17 Bridge. Read the report from NC DEQ.

NC DEQ Issues Advisory on Algal Blooms 6/11/19
Read the advisory.

Algal Bloom Update 6/1/19 - Little River
ARC&D Council member Rodney Johnson is reporting a 1,600 acre algal bloom in the Little River. Water samples collected mid-May by Rodney and his Green $aves Green team indicate that high amounts of phosphorus are entering the river through some of its tributaries.
Click to see a map of the bloom.

Algal Bloom Update 5/31/19 - Chowan River Near Colerain
Read the report from NC DEQ.

Algal Bloom Update 5/29/19
There have been reports of algal blooms on the Chowan River near Harrellsville, the boat basin on the Edenton waterfront, and some canals on the Little River. Also a small bloom on the east side of the Perquimans River. These localized blooms are in areas with slow water movement, generally on the east side of rivers where water and nutrients are pushed in by summer southwest winds. We have not heard so far of any extensive blooms in the Chowan River or Albemarle Sound.

Algal Bloom Update 5/16/19
Blooms were reported on the Chowan River near Harrellsville, and on the Perquimans River near Hertford. Read the report from the NC Division of Water Resources.

Chowan River at Wharf Landing 8/30/18. Read the report.

Chowan River at Colerain Update 8/14/18
Read the Division of Water Resources Algal Bloom Report

Chowan River at Wharf Landing and Edenton Bay Update 8/14/18
Read the Division of Water Resources Algal Bloom Report

April 20, 2017 by Catherine Clabby (https://www.northcarolinahealthnews.org/author/cclabby/)

With evidence that potentially dangerous algal blooms are becoming more common, a push is on to better understand and broadcast their dangers.

By Catherine Clabby

Harmful algal blooms, particularly the periodic explosive growth of cyanobacteria species, appear to be increasing in frequency and severity globally, researchers from the EPA (https://www.epa.gov/water-research/harmful-algal-blooms-drinking-water-treatment) and the Centers for Disease Control and Prevention (https://www.cdc.gov/habs/pdf/ohhabs-fact-sheet.pdf) say.

While most algae pose no risk, harmful algal blooms, whether green or reddish masses floating on the surface of water, can produce toxins (https://www.cdc.gov/habs/pdf/cyanobacteria_faq.pdf) that even in tiny amounts can harm people.

(https://1ibxcx1bcuig1rfxaq3rd6w9-wpengine.netdna-ssl.com/wp-content/uploads/2016/09/TaintedWatersLogo.jpg)Young children are likely at greatest risk (https://www.epa.gov/sites/production/files/2016-11/documents/harmful_algal_blooms_and_drinking_water_factsheet.pdf).

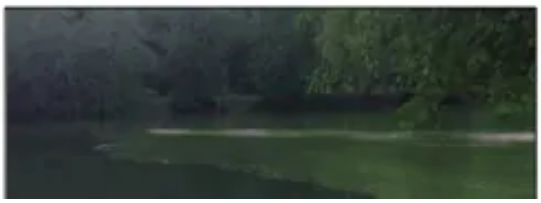
Swallowing or swimming in water during a dangerous bloom can produce (https://www.epa.gov/nutrientpollution/effects-human-health) skin rashes, liver injury and neurological changes. Toxic algae metabolites can also harm or kill pets and livestock.

The North Carolina Department of Environmental Quality confirmed the presence of potentially harmful cyanobacteria algae 19 times across North Carolina during the summer of 2016 alone.

How many potentially harmful blooms occurred in North Carolina in 2016 is unknown. State officials did detect cyanobacteria toxins called microcystin at five locations: Pamlico Sound, Albemarle Sound, a Brunswick County storm water pond, a Davie County pond and Greenfield Lake in New Hanover County, said Cobey Culton, a state Department of Health and Human Services spokesman.

The same uncertainty exists nationally. Blooms occur unpredictably and are not always documented, never mind assessed for their full chemical content.

“We need more data to inform prevention and mitigation of harmful-bloom health effects,” said Lorri Backer, a senior CDC environmental epidemiologist who spoke this week at a meeting of the North Carolina One Health Collaborative (http://nconehealthcollaborative.weebly.com/) at Research Triangle Park.



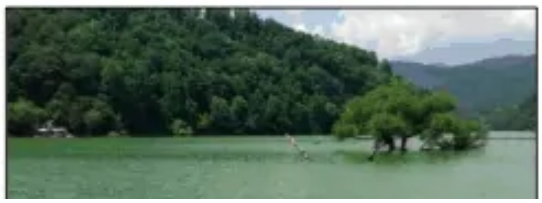


Figure 1. Beginning of the bloom in the Tuckasegee arm of Fontana lake

Figure 2. Photo of the bloom from the boat landing looking down toward Fontana Lake

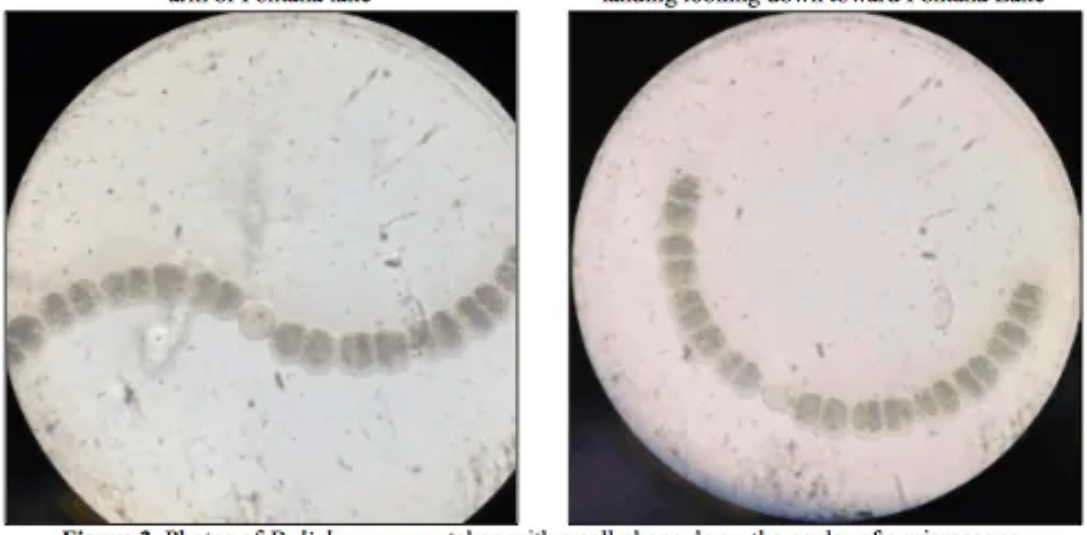


Figure 3. Photos of *Dolichospermum* taken with a cell phone down the ocular of a microscope

(https://1lbxcx1bcuig1rfxaq3rd6w9-wpengine.netdna-ssl.com/wp-content/uploads/2017/04/HAB_Tuckasegee.png)

A DEQ analysis of algae blooming in the Tuckasegee River where it reaches Lake Fontana last June tentatively identified the mass as a bloom of a Dolichospermum species, which may be capable of making toxins. Photos courtesy: NC Dept. of Environmental Quality

To achieve more clarity on this risk nationally, the CDC has launched a surveillance program, One Health Harmful Algal Bloom System (https://www.cdc.gov/habs/ohhabs.html). Public health programs in states, including North Carolina, report human and animal illness linked to algae exposure, as well as environmental conditions where the blooms occur.

The goal is to expand understanding and prevention of harmful blooms and illnesses associated with them.

Why now?

Scientists link evidence of increased incidences of toxic algal blooms to climate change. (https://www.epa.gov/sites/production/files/documents/climatehabs.pdf) Because algae have been around for

millions of years, they have adapted to live pretty much anywhere on this planet.

But they often thrive in warmer waters and as Earth's surface temperatures rise, so does that water.

Figure 1: Edenton Bay photograph from Chowan Edenton Environmental Group on July 18th

(https://1lbxcx1bcuig1rfxaq3rd6w9-wpengine.netdna-ssl.com/wp-content/uploads/2017/04/HAB_2.png)

DEQ's water sciences lab identified algae blooming in Edenton Bay off the Chown River last July as Dolichospermum planctonicum, a filamentous blue green algae that is capable of producing cyanotoxins. Source: NC DEQ

It doesn't help that human activities feed algae. Nitrogen and phosphorous that wash into our waterways from lawn and crop fertilizers, livestock animal manure, and underperforming sewage treatment plants are nutrients that feed algae.

North Carolina is no stranger to these blooms. They appear here mostly in freshwater ponds and sometimes quite dramatically in rivers, including the Chowan River (https://ncdenr.s3.amazonaws.com/s3fs-public/Water%20Quality/Environmental%20Sciences/FishKill/algae/Edenton%20Bay%20algae%20160719.pdf) last summer, said Mark Vander Borgh, a DEQ senior environmental biologist.

Sometimes they surge in estuaries as well, he said.

But they occur unpredictably, which can make them hard to catch, never mind quantify. Even as DEQ receives more complaints than ever, agency staff can't say conclusively that more are occurring, Vander Borgh said.

"We don't have the data to say if they are more common," he said.

Mina Shehee, of the North Carolina Division of Public Health, said after Backer's talk she wonders if small farmers raising livestock are losing more animals than they realize to toxins produced by algal blooms.

She's heard reports of animals dying for unknown reasons, she said, and wonders if ponds with algae might sometimes be to blame.

"We don't know the extent of it. They are out there but we're not capturing it," said Shehee, who works in the Occupational and Environmental Epidemiology Branch the Division.

She and colleagues are considering partnering with North Carolina Cooperative Extension agents to collect more data on animal deaths coinciding with algal blooms to get a clearer picture, she said.

Linking human, animal and environment

Spreading the word on the rising risk from harmful algal blooms fits well with the goals of the One Health movement, which strives to increase awareness (https://www.northcarolinahealthnews.org/2016/03/21/when-the-wild-and-tame-collide/) of ways that the health of people, animals and the environment overlap.

(https://1lbxcx1bcuig1rfxaq3rd6w9-wpengine.netdna-ssl.com/wp-content/uploads/2017/04/HAB_Nutrients.png)
When nutrients are plentiful and other conditions are right, microscopic algae, including Dolichospermum species, reproduce so profusely that they produce blooms that appear to coat ponds and other waters.

There is good evidence that pets have been sentinels to the presence of cyanotoxins.

While no human deaths have been linked to exposure to cyanobacteria in North Carolina, for instance, dogs have been made sick and may have perished (http://epi.publichealth.nc.gov/oee/algae/protect.html) from contact, according to state Department of Health and Human Services educational material on bloom threats.

"If water is sticky and green, people are not going to go into it. But a dog will," Rodgers said.

"If water is stinky and green, people are not going to go into it. But a dog will," Backer said.

Zach McKinney, a North Carolina State University graduate student in physiology attending the talk, said the general public is probably most in the dark about potential risks of harmful algae blooms.

While growing up in Pamlico County, he said, a member of the church his family attended had intense algae blooms strike ponds where he grew fish for profit. As the green slime spread, his fish were dying.

To help out, volunteers, including children, jumped into the water to scoop out the surviving fish to try to save them. That produced a lot of splashing and close contact with whatever the algae might have released into the air.

"No one considered the potential harm, (http://www.mdmag.com/journals/md-magazine-neurology/2016/march2016/does-blue-green-algae-cause-als-paul-cox-phd-is-on-a-global-quest)," he said.

Editor's note: *Locations where state officials detected toxic microcystins in 2016 were added to this story on April 22.*

About Catherine Clabby

Catherine Clabby (senior environmental reporter) is a writer and editor. A former senior editor at American Scientist magazine, Clabby won multiple awards reporting on science, medicine and higher education for the The Raleigh News & Observer. She is an alumna of the year-long Knight Science Journalism Program at MIT. Contact: catherine.clabby@gmail.com