Report of the NJDEP Science Advisory Board

Prepared by the Science Advisory Board
Ecological Processes Standing Committee

DREDGING: A REVIEW OF THE DOCUMENTED IMPACTS OF DREDGING ON THE COASTAL ECOSYSTEMS OF NEW JERSEY

Approved by the
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The following report has been issued by the Science Advisory Board to the Commissioner of the New Jersey Department of Environmental Protection

Response to the Charge Question regarding:

The impacts of dredging and dredged material management activities on New Jersey coastal ecosystems and taxa, as well as potential mitigation options for such impacts.

This report was prepared by the Ecological Processes Standing Committee and sent to the Science Advisory Board for review and approval. The Science Advisory Board based this final report on those recommendations from the Ecological Processes Standing Committee.

Members of the Ecological Processes Standing Committee include:

Chair – Mr. Charles R. Harman
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DEDICATION

This manuscript is dedicated to the memory of the Former Chair of the Ecological Processes Standing Committee, the late Dr. Carolyn Bentivegna of Seton Hall University. Your untimely passing is a loss to science and to those of us who knew you. You will be missed, and always remembered.

ACKNOWLEDGEMENTS

The members of the Ecological Processes Standing Committee (EPSC) would like to thank the NJDEP staff for their support and assistance in the preparation of this report. We also thank the USGS for the logistical support at their Lawrenceville facility.

EXECUTIVE SUMMARY

The New Jersey Department of Environmental Protection (NJDEP) is charged with evaluating applications for dredging projects in coastal areas of the State, and issuing (or denying) permits for those projects pursuant to the Coastal Zone Management Rules (N.J.A.C. 7:7). As a component of the permit review, the NJDEP has identified questions related to the timing issues of when permitted dredging actions can occur; answers to which may help to improve the application process, or the development of guidance to the regulated community as to ways in which to better prepare for the project. The overall concern is that the presence of multiple layers of timing restrictions of when dredging can occur can make dredging operations challenging to schedule and finance. The Science Advisory Board, Ecological Processes Standing Committee (EPSC) was tasked with documenting the impacts of dredging and dredged material management activities on coastal ecosystems and taxa, with emphasis on those ecosystems and taxa present in NJ, and potential mitigation options (such as minimization, avoidance, timing restrictions, restoration, and dredging techniques) for such impacts. This information would then be summarized in a report of findings.

Dredging is a critical tool in the maintenance of our nation's waterways. In both freshwater and salt water settings, dredging is used to clear shoals from navigational channels, allowing for the safe passage of ships and shipping; removal of accumulated sediment from harbors, piers, and marinas; and movement and deposition of sand in beach areas as a means of beach replenishment. Additionally, environmental dredging is one of the primary means for the removal of sediment contaminated with chemicals of concern such as metals and organic compounds released from hazardous waste sites into waterways.

Dredging activities can have both direct and indirect impacts on the environment. Direct environmental impacts can include mortality to benthic organisms, fish or other fauna from the physical removal of sediment through dredging, or the turbidity resulting from dredging operations, as well as contaminants which are mobilized into the water column, noise and disturbance to aquatic biota from the dredge and any supporting vessels.

Indirect environmental impacts include the potential for increased shoreline erosion in some areas, loss of benthic habitat (particularly when sediment and more sessile biota are removed down to bedrock), associated loss of prey to higher organisms such as fish and birds, long term impacts on water quality as a result of removing bivalves that filter nutrients or fine sediments that adsorb contaminants, impacts to populations of migratory fish, reduction in spawning sites, and long term effects on species/community composition as a result of altering bathymetry.

Timing restrictions are employed to reduce the risk of impact by project activities to sensitive life stages of some federal- and state-managed species and habitats. Impacts from dredging include entrainment, sedimentation, potential contamination, disturbance from sound impacts, and turbidity impacts. Recommendations for timing restrictions provided by resource agencies are important for the conservation of state and federal resources, however the resultant windows of time available for work often limit the time available for dredging projects to be accomplished.

The Standing Committee identified a number of specific recommendations relative to dredging and the posed charge question.

1. INTRODUCTION

Dredging is a critical tool in the maintenance of our nation's waterways. In both freshwater and saltwater settings, dredging is used to clear shoals from navigational channels, allowing for the safe passage of ships and shipping; removal of accumulated sediment from harbors, piers, and marinas; and movement and deposition of sand in beach areas as a means of beach replenishment. Additionally, environmental dredging is one of the primary means for the removal of sediment contaminated with chemicals of concern such as metals and organic compounds released from hazardous waste sites into waterways.

However, dredging does have its limitations. Concerns associated with dredging can include:

1. Impacts to sediment-dwelling fish and shellfish associated with the removal of the upper layers of sediment in which they live;
2. Increases in turbidity in the water column resulting in the exposure of pelagic aquatic species to sediment-borne contamination or the smothering of eggs or benthic organisms or submerged aquatic vegetation (SAV) beds downstream of a dredging site;
3. Increased noise levels resulting in disturbance to avifauna during breeding periods; and
4. Increased underwater noise levels disturbing migration and spawning periods for fish

Charge

The New Jersey Department of Environmental Protection (NJDEP) is charged with evaluating applications for dredging projects in coastal areas of the State, and issuing (or denying) permits for those projects pursuant to the Coastal Zone Management Rules (N.J.A.C. 7:7). As a component of the permit review, the NJDEP has identified questions related to the timing issues of when permitted dredging actions can occur; answers to which may help to improve the application process, or the development of guidance to the regulated community as to ways in which to better prepare for the project. The overall concern is that the presence of multiple layers of timing restrictions of when dredging can occur can make dredging operations challenging to schedule and finance. The Science Advisory Board, Ecological Processes Standing Committee (EPSC) was tasked with documenting the impacts of dredging and dredged material management activities on coastal ecosystems and taxa, with emphasis on those ecosystems and taxa present in NJ, and potential mitigation options (such as minimization, avoidance, timing restrictions, restoration, and dredging techniques) for such impacts. This information would then be summarized in a report of findings.

Following are the findings of the EPSC with respect to the charge.

2. REGULATORY FRAMEWORK

Dredging is the removal of bottom soils or sediments from wetlands or State open water through the use of mechanical, hydraulic, or pneumatic tools in an effort to restore or maintain original bottom contours of waterbodies. Dredging activities are regulated by the Department's Office of Dredging and Sediment Technology within the Division of Land Use Regulation. In coastal waters, dredging is regulated by the Coastal Zone Management rules at N.J.A.C. 7:7, and is defined as the removal of sediment located waterward of the spring high water line. Dredging does not

include excavation. Dredging activities may be authorized under a general permit or through the issuance of an individual permit depending upon the scope and degree of impacts.

In evaluating the environmental impacts of dredging, it is important to understand the regulatory context by which both the State of New Jersey and the federal government regulate the use of dredging in both freshwater and in coastal waters, and the control of their impact. This is accomplished through the imposition of conditions on permits granted, particularly regarding "dredging windows" designed to protect fisheries and other natural resources potentially impacted by dredging projects. NJDFW (2008) outlines how land use permits (including dredging permits) are to be processed in terms of the protection of fish and wildlife. The regulatory framework imposes important bounds on the charge question regarding dredging impacts, since currently impacts to birds, wildlife and invertebrates may not be directly regulated.

In addition to state regulations, any entity that proposes construction or fill activities in waters of the United States, including wetlands, must obtain a permit from the U.S. Army Corps of Engineers (USACE). Activities within New Jersey generally fall under the jurisdictional authority of either the New York District (e.g., New York/New Jersey Harbor area, including Raritan Bay and the Raritan River, the Hackensack and Passaic Rivers and their watersheds, Newark Bay, the Arthur Kill, etc.), or the Philadelphia District (Delaware River, coastal areas of southern New Jersey). At the Federal level, the USACE has jurisdiction over all construction activities in tidal and/or navigable waters under Section 404 of the Clean Water Act, and Section 10 of the Rivers and Harbors Act, including adjacent wetlands, shoreward to the mean high-water line. In other areas such as non-tidal waterways, adjacent wetlands, isolated wetlands, forested wetlands, and lakes, the USACE has regulatory authority over the discharge of dredged or fill material.

The following major regulations currently act to control the environmental impacts of dredging:

- 33 CFR Part 323 Permits for Discharges of Dredged or Fill Material into Waters of the United States. These regulations provide authority to USACE for implementing the nationwide permitting program. In issuing permits USACE must confer with other agencies in evaluating impacts, include those on fisheries.

The USACE regulates impacts from dredging activities through the nationwide permit program. Generally, if an activity does not meet the requirements of any single permit under the nationwide general permit program, then an individual permit would be required.

Coastal Zone Management Consistency Determination

Section 307(c)(1) of the Coastal Zone Management Act (CZMA) requires the USACE to provide a consistency determination and receive state agreement prior to the issuance, reissuance, or expansion of activities authorized by a nationwide permit that authorizes activities within a state with a Federally-approved Coastal Management Program when activities that would occur within, or outside, that state's coastal zone will affect land or water uses or natural resources of the state's coastal zone.

Endangered Species- Section 7 of the Endangered Species Act

The Endangered Species Act (ESA) was passed to protect and recover species identified as threatened and or endangered (T&E species) from becoming extinct, as well as protect the habitat upon which they are dependent. The ESA is administered by the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS) located in the Department of Commerce. The USFWS has primary responsibility for terrestrial and freshwater organisms, and NMFS has responsibility for marine wildlife such as whales and anadromous fish.

From a dredging perspective, no activity is authorized if that activity is likely to jeopardize the continued existence of a threatened or endangered species as listed or proposed for listing under the ESA, or to destroy or adversely modify the critical habitat of listed species. In certain instances, environmental windows have been identified as management tools to protect sensitive life stages of endangered species. An example of this is the Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). General dredging windows for this species are from March 1 to June 30. While the USACE has developed a not likely to adversely affect (NLAA) process by which the Corps can work with the NMFS to potentially modify the dredging windows for an ESA-listed species, it is not a simple process and one that is rarely pursued.

Magnuson Stevens Fishery Conservation and Management Act (MSA)

The Magnuson-Stevens Fishery Conservation and Management Act (MSFCMA), which was reauthorized and amended by the Sustainable Fisheries Act (1996), requires the eight regional fishery management councils to describe and identify essential fish habitat (EFH) in their respective regions, to specify actions to conserve and enhance that EFH, and to minimize the adverse effects of fishing on EFH. Congress defined EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity." The MSFCMA requires the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service to assist the regional fishery management councils in the implementation of EFH in their respective fishery management plans.

The Northeast Fisheries Science Center compiled the available information on the distribution, abundance, and habitat requirements for each of the species managed by the New England and Mid-Atlantic Fishery Management Councils in a series of EFH species reports. The EFH species reports comprise a survey of the important literature as well as original analyses of fishery-independent data sets from the NOAA Fisheries Service and several coastal states. The species reports are also the source for the current EFH designations by the New England Fisheries Management Council (NEFMC) and Mid-Atlantic Fisheries Management Council (MAFMC), and have understandably begun to be referred to as the "EFH source documents." The EFH source documents for managed species can be found at https://www.nefsc.noaa.gov/nefsc/habitat/efh/. Species managed under MSA through the NEFMC and the MAFMC for New Jersey are listed in Table 1.

Table 1: Essential Fish Habitat Managed Species

Species	Scientific Name
Vertebrates	
American plaice	*Hippoglossoides platessoides*
Atlantic angel shark	*Squatina dumerili*
Atlantic butterfish	*Peprilus triacanthus*
Atlantic cod	*Gadus morhua*
Atlantic mackerel	*Scomber scombrus*
Atlantic. sharpnose shark	*Rhizopriondon terraenovae*
Atlantic sea herring	*Clupea harengus*
Black sea bass	*Centropristis striata*
Bluefish	*Pomatomus saltatrix*
Clearnose skate	*Raja eglanteria*
Cobia	*Rachycentron canadum*
Dusky shark	*Carcharhinus obscurus*
Haddock	*Melanogrammus aeglefinus*
King mackerel	*Scomberomorus cavalla*
Little skate	*Leucoraja erinacea*
Monkfish	*Lophius americanus*
Ocean pout	*Macrozoarces americanus*
Pollock	*Pollachius virens*
Redfish	*Sebastes fasciatus*
Red hake	*Urophycis chuss*
Sandbar shark	*Carcharhinus plumbeus*
Sand tiger shark	*Carcharias taurus*
Scalloped hammerhead shark	*Sphyrna lewini*
Scup	*Stenotomus chrysops*
Spanish mackerel	*Scomberomorus maculatus*
Spiny dogfish	*Squalus acanthias*

Species	Scientific Name
Summer flounder	*Paralichthys dentatus*
Tilefish	*Lopholatilus chamaeleonticeps*
Tiger shark	*Galeocerdo cuvieri*
White hake	*Urophycis tenuis*
Whiting	*Merluccius bilinearis*
Windowpane flounder	*Scophthalmus aquosus*
Winter flounder	*Pseudopleuronectes americanus*
Winter skate	*Leucoraja ocellata*
Witch flounder	*Glyptocephalus cynoglossus*
Yellowtail flounder	*Limanda ferruginea*
Invertebrates	
Atlantic sea scallop	*Placopecten magellanicus*
Long finned squid	*Loligo pealeii*
Ocean quahog	*Artica islandica*
Surf clam	*Spisula solidissima*

Source: https://www.nefsc.noaa.gov/nefsc/habitat/efh/

The NOAA National Marine Fisheries Service (NMFS) under the MSA requires federal agencies such as the USACE to consult with NMFS on projects, including dredging projects that may adversely affect EFH. Any proposed activity (for which a nationwide permit is being applied) that is proposed within 50 feet of submerged aquatic vegetation (SAV) beds, mapped SAV habitat and/or within fisheries Habitat Areas of Particular Concern (HAPC) as depicted by the Essential Fish Habitat Mapper (http://www.habitat.noaa.gov/protection/efh/efhmapper/) must file a pre-construction notification so that agencies are aware of potential impacts.

Other Laws and Regulations

Contaminants released into the water column by dredging are regulated by Section 401 of the Clean Water Act but may be difficult to measure and monitor in tidal situations. Contaminant release is a function in part of grain size of the areas being dredged and is generally more of a concern in fine-grained sediments such as the New York/New Jersey Harbor estuary and the Delaware River north of Trenton.

There are several other laws and regulations that may govern dredging activities, including the Fish and Wildlife Coordination Act, Executive Orders, Memorandums, such as EO 11988, Floodplain Management, EO 11990, Protection of Wetlands, EO1211, Environmental Effects of Major Federal Actions, EO 12989, Environmental Justice in Minority Populations and Low-Income Populations. These regulations may impact dredging activities on a case by case basis and their descriptions would be outside the scope of this report.

3. METHODS OF DREDGING

There are many different methods utilized for dredging in the United States. Each of the methods has distinct advantages and disadvantages, with the choice of equipment depending upon the following factors (USACE 0215):

1. Physical characteristics of the material being dredged;
2. Quantities and distribution of the material being dredged;
3. Depth of the material to be dredged;
4. Location of both the dredging and placement sites and the distance between them;
5. Physical environment of the dredging and placement areas, as well as the area in between;
6. Concentrations of contaminants within the sediment to be dredged;
7. Methods to be used in placing the sediment after dredging;
8. Production rates that are required for the project; and
9. Types of dredges that are available for the project.

Methods of dredging are important, since they may vary regarding their degree and type of impact. In general, there are two categories of dredges available for dredging projects: hydraulic dredges and mechanical dredges. Hydraulic dredges are characterized by the use of a centrifugal pump to dredge sediment and transport it as a liquid slurry to the placement area. The common types of hydraulic dredges include hopper dredges and cutterhead pipeline dredges, with dustpan dredges and sidecaster dredges used on a limited basis (Herbich 2000 and USACE 2015). Mechanical dredges are characterized by the use of some form of bucket to excavate and raise the bottom material. Mechanical dredges can be further divided into two subgroups based on how there buckets are connected to the dredge. Clamshell and dragline dredges have their buckets connected to the dredge by a wire rope, while backhoe dredges have their buckets connected structurally to the dredge (USACE 2015).

A further description of the most commonly used dredging methods in the United States, their advantages and disadvantages is as follows:

Hydraulic Pipeline Cutterhead Dredges

The hydraulic pipeline cutterhead dredge, or cutterhead dredge is the most commonly used dredging vessel and is generally the most efficient and versatile. It performs the major portion of the dredging workload in the USACE dredging program. Because it is equipped with a rotating cutter apparatus surrounding the intake end of the suction pipe, it can efficiently dig and pump all types of alluvial materials and compacted deposits. A cutterhead dredge can excavate a wide range of materials, including clay, silt, sand, and gravel. The cutterhead dredge is suitable for maintaining

harbors, canals, and outlet channels where wave heights are not excessive. A cutterhead dredge designed to operate in calm water does not effectively operate offshore in waves over 2-3 feet in height (USACE 2015).

Advantages (USACE 2015)

- Capable of excavating most types of material and pumping it through pipelines for long distances to upland placement sites;
- Results in a maximum of efficiency and productivity; and
- Larger and more powerful cutterhead dredges are able to go through rocklike formations without blasting.

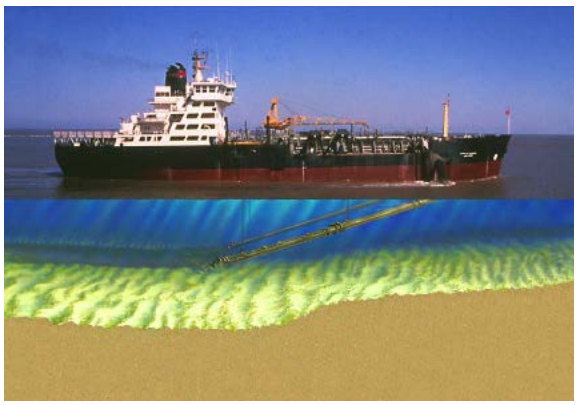
Source: USACE 2015

Limitations (USACE 2015)

- Limited capacity for working in open-water areas without posing a danger to personnel and equipment;
- Draft of the dredge may make it challenging to operate in shallower waters outside of the channel; and
- Generally not self-propelled and require large towboats to move them between dredging locations.

Hopper Dredges

Hopper dredges are self-propelled seagoing ships that vary in length from 180 to 510 feet, and have the hulls and lines of ocean vessels. They were developed for maintenance work and are equipped with propulsion machinery, sediment containers (hoppers), dredge pumps, and other special equipment required to perform their essential function of removing material from a channel bottom or ocean bed and placing that material in open water or upland sites. During dredging operations, hopper dredges travel at a ground speed of 2-3 miles per hour and can dredge in depths from approximately 10-140 feet Hopper dredges are equipped with high-volume, low-head dredge pumps, and dredging is accomplished by progressive traverses over the area to be dredged. In addition, they are able to maintain the velocities required to carry a high percentage of solids in the suction and discharge lines. The drag head is moved along the channel bottom as the vessel moves forward at speeds up to 3 miles per hour (5 kilometers per hour). The



Source: USACE 2015

dredged material is entrained into the drag head, moves up the drag pipe, and deposited and stored in the hoppers of the vessel. Hopper dredges are used mainly for maintenance dredging in exposed harbors and shipping channels where traffic and operating conditions rule out the use of stationary dredges. The materials excavated by hopper dredges cover a wide range of types, but hopper dredges are most effective in the removal of material that forms shoals after the initial dredging is completed (USACE 2015).

Advantages (USACE 2015)

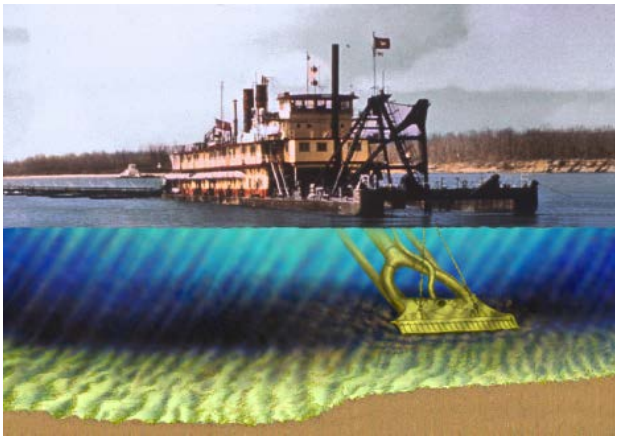
- They can work safely in rough, open water;
- They can move under their own power; and
- They are the most economical type of dredge when placement areas are not within economic pumping distance.

Limitations (USACE 2015)

- Their deep draft preclude their use in shallow areas;
- They operate with less precision than other types of dredges; and
- Consolidated and cohesive clay material cannot be dredged using this equipment.

Dustpan dredges

The dustpan dredge is a self-propelled hydraulic pipeline dredge that uses a widely flared dredging head along which are mounted pressure water jets. The jets loosen and agitate the sediments, which are then captured in the dustpan head as the dredge itself is winched forward into the excavation. The piping system used with this dredge make it effective only in rivers and sheltered waters. It cannot be used in estuaries or bays that have significant wave action. Because of a lack of cutterhead to loosen hard, compact material, this type of dredge is suitable for only high volume, loose material (USACE 2015).



Source: USACE 2015

Advantages (USACE 2015)

- It is self-propelled; and
- Has a high production rate.

Limitations (USACE 2015)

- Only works well with loose materials such as sand or gravel.

Sidecasting Dredges

The sidecasting type of dredge is a shallow-draft seagoing vessel, specially designed to remove material from the bar channels of small coastal inlets. The hull design is similar to that of a hopper

dredge; however, sidecasting dredges do not usually have hopper bins. Instead of collecting the material in hoppers onboard the vessel, the side-casting dredge pumps the dredged material directly overboard through an elevated discharge boom; thus, its shallow draft is unchanged as it constructs or maintains a channel. The sidecasting dredge picks up the bottom material through two drag arms and pumps it through a discharge pipe supported by a discharge boom. During the dredging process, the vessel travels along the entire length of the shoaled area, casting material away from and beyond the channel prism (USACE 2015).

Source: USACE 2015

Advantages (USACE 2015)

- It is self-propelled; and
- It can move quickly from one location to the next.

Limitations (USACE 2015)

- Productivity can be affected by water levels in tidally influenced areas, with operations at high tide being the most productive.

Bucket Dredges

Bucket dredges are a type of mechanical dredge that use a grabbing device or "bucket" to excavate the material being dredged. There are different types of buckets depending upon the needs of the dredging project. Material that is dredged is generally placed in a scow or hopper barge to be towed to a placement area. In some instances, the dredged material can be sidecast. Bucket dredge production is a function of both the loading (excavation) and hauling components of its operation. Production rates while loading depend on several factors: the size and weight of the bucket, the operating characteristics of the bucket, the type of material to be excavated, the face thickness or bank height of the material, operator efficiency, and the bucket cycle time. Operating characteristics affect the bucket's fill factor (the decimal equivalent of the percent volume of bucket actually filled) and include the bucket weight, bucket shape, closing edge configuration (toothed or smooth), and closing action. A complete bucket cycle time is defined as the time required to lower the open bucket to the bottom, mechanically grab the material, close the

Source: USACE 2015

bucket, and then raise, position, and release the bucket either over a waiting scow or to the side (sidecasting).

Bucket dredges may be used to excavate most types of materials except the most cohesive consolidated sediments and solid rock. Bucket dredges usually excavate a heaped bucket of material, but during hoisting, turbulence washes away part of the load. Once the bucket clears the water surface, additional losses may occur through rapid drainage of entrapped water and slumping of the material heaped above the rim. Loss of material is also influenced by the fit and condition of the bucket, the hoisting speed, and the properties of the sediment. Even under ideal conditions, substantial losses of loose and fine sediments usually occur. Because of this, the bucket dredge may employ special buckets if it is being considered for use in dredging applications requiring reduced sedimentation resuspension rates. To minimize the turbidity generated by a bucket dredge operation, enclosed buckets (clamshells) have been developed for navigation and environmental dredging projects.

Mechanical dredges (bucket and clamshell) are also associated with higher suspended sediment concentrations than hydraulic (hopper and cutterhead) methods. Mechanical dredges generate suspended sediments through both the impact of the bucket on the bottom and the withdrawal from the bottom, washing of material out of the bucket as it moves through the water column and above the water surface as well as additional dredged material loss when the barge is loaded. (Tavolaro et al. 2007 and USACE 2015).

Advantages (USACE 2015)

- It can remove bottom materials consisting of clay, hard-packed sand, glacial till, stone, or blasted rock material;
- Bucket dredges require less room to work than most other types of dredges; and
- The excavation is precisely contained.

Limitations (USACE 2015)

- Difficult to retain soft, semi-suspended fine-grained materials in conventional buckets;
- Barges are required to move the material to a placement area; and
- Production is relatively low compared with the production of cutterhead and dustpan dredges.

Backhoe Dredges

The backhoe dredge uses a bucket that is structurally connected to the dredge by the rigid member configuration as shown in Figure 2-36. To increase digging power, the dredge barge is moored on powered spuds that transfer the weight of the forward section of the dredge to the bottom to provide reaction forces to the digging-induced forces. The maximum bucket size that can be used for a specific project depends on the rated capacity of the excavator, sediment characteristics, and water depth. Larger backhoes can excavate to a maximum depth of approximately 80 feet. Because the bucket of a backhoe is connected by rigid structural members, more force can be applied to it, allowing these types of dredges to work in “harder” materials (relatively cohesive consolidated materials, weak rock, and debris) than can cable-connected buckets. Backhoe operational

characteristics provide relatively high excavation accuracy, and they can work closely around structures. The density of sediment excavated can almost equal its *in situ* density but, like other conventional mechanical dredges, it may generate a relatively large amount of sediment resuspension at the dredge site. The best use of the backhoe dredge is for excavating hard, compacted materials, rock, or other solid materials after blasting. Although it can be used to remove most bottom sediments, the violent action of this type of equipment may cause considerable sediment disturbance and resuspension during maintenance digging of fine-grained material. In addition, a significant loss of fine-grained material occurs from the bucket during the hoisting process.

Source: USACE 2015

The backhoe dredge is most effective around bridges, docks, wharves, pipelines, piers, or breakwater structures because it does not require much area to maneuver; there is little danger of damaging the structures since the dredging process can be controlled accurately. No provision is made for dredged material containment or transport, so the backhoe dredge must work alongside the placement area or be accompanied by barges during the dredging operation (USACE 2015).

Advantages (USACE 2015)

- It can remove almost any type of sediment material;
- The power of the dredge makes it optimal for the removal of hard and compact materials;
- It can be used to remove old piers, breakwaters, foundations, piling, roots, stumps and other obstructions;
- It requires less room to work than hydraulic dredges; and
- Its excavation can be precisely controlled.

Limitations (USACE 2015)

- Difficult to retain soft, semi-suspended fine-grained materials in the bucket of a backhoe dredge;
- Barges are required to move the dredged material to a placement area;
- The production is relatively low compared with the production of cutterhead and dustpan dredges; and
- As with the clamshell dredge, this mechanical dredge is associated with higher suspended sediment concentrations than hydraulic (hopper and cutterhead) methods.

4. IMPACTS OF DREDGING

Dredging activities can have both direct and indirect impacts on the environment. Each of these impacts are influenced by several factors, including:

1. The nature of the environment and associated natural resources in the area being dredged;
2. The nature of the surrounding habitat (e.g., whether it is undisturbed salt marsh or a bulkheaded urban waterfront)
3. Dredging project characteristics such as vertical and horizontal extent of the area dredged (e.g., whether it is initial deepening or maintenance dredging of what has silted into the channel), the substrate characteristics (e.g., physicochemical characteristics of the material being removed), degree and type of sediment contamination present, hydrogeomorphic characteristics of the water body being dredged, mode of dredging, geometry of the dredge cut; and,
4. Ancillary impacts associated with dredging activities such as noise, diesel emissions, and secondary impacts are related to the size and type of dredge, duration of the dredging project;
5. Method of disposal, amount and location of dewatering and choice of disposal sites may affect the adjacent terrestrial and aquatic environments.

Direct environmental impacts can include mortality to benthic organisms, fish or other fauna, and submerged aquatic vegetation (SAV) from the physical removal of sediment through dredging, or the turbidity resulting from dredging operations. Dredging can also cause contaminants to be mobilized into the water column, as well as noise and disturbance to aquatic biota from the dredge equipment and any supporting vessels.

Indirect environmental impacts include the potential for increased shoreline erosion in some areas, loss of benthic habitat (particularly when sediment and more sessile biota are removed down to bedrock, associated loss of prey to higher organisms such as fish and birds, long term impacts on water quality as a result of removing bivalves that filter nutrients or fine sediments that adsorb contaminants, impacts to populations of migratory fish, reduction in spawning sites, and long term effects on species/community composition as a result of altering bathymetry, and loss of SAV beds which serve as important foraging areas and refugia at critical life stages for numerous species).

From a very general standpoint, hydraulic dredges are seen to have less environmental impacts that bucket-type mechanical dredges. That is primarily based on the fact that the use of hydraulic dredges results in much less sediment being suspended in the water column.

4.1 Review of Wenger et al. (2017)

Wenger et al (2017) reviewed the impacts of suspended sediments on fish. While the entire paper is attached as Appendix A, the following section summarizes the findings of the paper.

Overall effects

Mortality is an important endpoint, but detection of early stress is important because it will allow intervention before mortality occurs.

Contaminated sediments have greater negative effects than clean sediments since toxicity combines with the other stresses. Lethal effects on eggs and larvae are greater than on older life history stages.

Suspended sediments

Suspended sediments affect behavior, physiology, and mortality of fish and benthic organisms, and can cause physical damage. The degree of effects depends on the concentration of suspended sediments, duration of exposure, life history stage, and type of response. With higher concentrations and longer exposure, effects are more severe, as expected. The settling rate of suspended sediments depends on particle size.

Behavioral responses –avoidance is induced by low levels of oxygen, which can lead to community shifts when organisms move elsewhere (Collin and Hart, 2015; DeJonge et al. 1993). Recovery depends on the availability of suitable habitat and the plasticity of the particular species (Pirotta et al. 2013). Turbidity causes light attenuation (Jones et al., 2015) which will decrease vision and activities dependent on vision, such as larvae finding appropriate habitat, and foraging (O'Connor et al., 2015; Wenger et al., 2011; Wilson et al 2008). Suspended sediments can decrease foraging in planktivorous and piscivorous fishes (Utne-Palm, 2002; De Robertis et al., 2003) as well as feeding of herbivores on algae if sediments settle there (Bellwood and Fulton, 2008). Reductions in feeding can lead to growth reductions, worsening of biological condition, and limitations of reproductive output (Sweka and Hartman, 2001; Wenger et al., 2012).

Physiological responses –Suspended sediments can damage gill membranes (Au et al., 2004; Hess et al., 2015). This physical damage can cause an increase in pathogens and parasites (Lowe et al., 2015), as well as reduced respiration, nitrogen excretion, and ion exchange (Appleby and Scarratt, 1989; Au et al., 2004; Wong et al., 2013). The greatest effects are seen in larvae and juvenile fishes, which have smaller gills that are more easily clogged and abraded by suspended sediment (Appleby and Scarratt, 1989). Since fish larvae typically have higher oxygen requirements, reduced uptake efficiency can easily lead to reduced growth and increased mortality (Nilsson et al., 2007).

Released Contaminants

Common contaminants on sediments include metals, PAHs, and PCBs. Multiple contaminants have physiological impacts on reproduction (steroidogenesis, vitellogenesis, gamete production, and spawning) development, and behavior (Johnson et al., 2014). The dredging process can cause some hydrophobic organic chemicals to desorb from the sediments and be more available for bioaccumulation (Bridges et al., 2008; Steuer, 2000). Metals can also be released from sediments, some metals more easily than others (Maddock et al., 2007), and cause effects on reproduction, development, osmoregulation, and hormones (Chow and Chang 2003; Kime et al., 1996; Hutchinson et al., 1994).

Entrainment

The direct uptake of living things into dredging equipment results in localized by-catch. The direct removal of eggs and larvae is typically lethal. When entrainment occurs near dense patches of eggs

and larvae of fish and benthic organisms, population reductions could ensue. Dredging operations can avoid entrainment by setting time windows and avoiding nursery areas (Suedel et al., 2008). For mobile juveniles and adults entrainment causes less mortality, but effects will depend on the method of dredging and the mode and site of deposition of the sediments. For example, mortality rate of estuarine fish in Washington after hydraulic dredging was 38% but was 60% for pipeline dredges with a cutter head (Armstrong et al., 1982).

Noise

Sound from dredging generally is not lethal, but causes behavioral and physiological changes which are greatest in fish with a swim bladder that is used in hearing (Popper et al., 2014). Impacts of anthropogenic sound include behavioral responses, masking, stress and physiological responses, damage to auditory tissue and hearing loss, structural and cellular damage to non-auditory tissues, impairment or lateral line functions, and abrasion of eggs and larvae caused by particle motion (Popper and Hastings, 2009; Popper et al., 2014). While there have been many studies on effects of sound on fishes (reviewed by Hawkins et al., 2015 and Popper and Hastings, 2009), there have been few studies specifically on sounds caused by dredging operations. It may cause temporary hearing loss in fishes and increase in stress-related cortisol. Studies of dredging noise by Reine et al. (2014) and Thomson et al. (2009) indicate that it produces lower levels of sound, generally below 1 kHz. Effects vary according to bathymetry, and are likely to include temporary hearing loss in some fish species, behavioral effects, and increased stress hormones. Dredging noise may mask natural sounds used by fish and other aquatic species to locate suitable habitat (Simpson et al., 2005)

Recommendations and summary of Wenger et al. (2017)

The literature to date has focused mostly on effects of suspended sediments. There is a need for more study on the other types of stresses caused by dredging, such as noise, contaminated sediments, abrasion and entrainment. There is a strong need for more field studies (*in situ*). The combination of toxicity, chronic stress, loss of habitat, and repeated exposure to multiple contaminants and pulses of suspended sediments can result in synergistic effects. Since larvae and eggs of fishes and other aquatic species are most vulnerable to dredging operations, windows should be implemented to avoid stressing these early life stages.

Indirect effects, such as loss of prey and habitat changes, are also in need of study. A BACI design (Before-After-Control-Impact) is appropriate for field studies to assess responses of targeted life history stages to multiple stresses over time.

4.2 Chemical/physical impacts

Chemical and physical impacts include suspension of sediment particles into the water column, burial of aquatic invertebrates elsewhere from settling of suspended particles (Jones, 1991; Messieh et al. 1991), or bank erosion, and alteration of the flow patterns within the water body being dredged, ultimately altering habitat Emerson, 1971). Physical alteration of the bottom by sediment removal can result in changes in water flow, particularly in rivers and estuaries (Newell et al. 1998). This can result in eventual redistribution of sediment and a change in depositional patterns that may impact habitat, particularly those areas important to sessile organisms. Large

scale dredging projects (e.g., Delaware River channel deepening) have the potential to modify salinity patterns within estuarine systems (Quammen and Onuf 1993; Chubarenko and Tchepikova 2001). For example, a deeper channel may allow more saline water to travel upstream during an incoming tide, affecting species distribution, particularly during drought periods.

4.3 Other Studies on Noise

Wale et al. (2013) investigated behavioral responses of shore crabs to ship noise, the most common source of underwater noise. This noise affected foraging and antipredator behavior in the green crab, *Carcinus maenas*. Ship noise playback was more likely than ambient-noise playback to disrupt feeding, although crabs in the two sound treatments had equal speed at finding food. While crabs exposed to ship noise playback were just as likely as ambient-noise controls to detect and respond to a simulated predatory attack, they were slower to retreat to shelter. These results demonstrated that anthropogenic noise has the potential to increase the risks of both starvation and predation in crabs. Being distracted by noise also made hermit crabs more vulnerable to predation. When exposed to boat motor playback, the hermit crab *Coenobita clypeatus* allowed a simulated predator to approach closer than usual before initiating the normal hiding response (Chan et al., 2010).

Exposure of the lobster *Palinurus elephas* to boat noise caused significant changes in locomotor behaviors (Filiciotto et al., 2014). Exposed lobsters significantly increased their locomotor activities. They also exhibited altered their hemolymphatic parameters, with elevated indicators of stressful conditions, such as glucose and total proteins.

Solan et al. (2016) exposed three sediment-dwelling species – the langoustine (*Nephrops norvegicus*), the Manila clam (*Ruditapes philippinarum*) and the brittlestar (*Amphiura filfiformis*) to either continuous broadband noise (CBN) similar to shipping traffic and intermittent broadband noise (IBN) similar to marine construction activity. They found that exposure to underwater broadband sound fields that resemble offshore shipping and construction activity altered their sediment processing behavior. The langoustine, which disturbs the sediment to create burrows reduced the depth of sediment redistribution when exposed to either IBN or CBN. Under CBN and IBN there was evidence that bioirrigation, which is strongly influenced by the activity of the siphon, increased. The Manila clam that lives in the sediment and connects to the overlying water through a retractable siphon, reduced its surface activity under CBN. Bioirrigation was greatly reduced by CBN and slightly reduced by IBN. In contrast, the exposure had little impact on the brittlestar's behavior.

Some fish communicate using sounds. Vasconcelos et al. (2007) investigated the effects of ship noise on the detection of conspecific vocalizations by the Lusitanian toadfish, *Halobatrachus didactylus*. Ambient and ferry boat noises were recorded, as well as toadfish sounds. Hearing was measured under quiet conditions in the laboratory and in the presence of these noises at levels found in the field. In the presence of ship noise, auditory thresholds increased considerably because the boat noise was within the most sensitive hearing range of this species. The ship noise decreased the fish's ability to detect conspecific sounds, which are important in agonistic behaviors (e.g., displays, retreats, placation, and conciliation) and mate attraction.

Sebastianutto et al. (2011) evaluated the effects of boat noise on the goby, *Gobius cruentatus,* which normally emit sounds during agonistic encounters. They played a field-recorded diesel engine boat noise during aggressive encounters between an intruder and a resident fish in a laboratory. Agonistic behavior of the residents was reduced by the noise: they were more submissive and won fewer encounters. The authors suggested that sound production is used for territorial defense, and since it was impaired by the boat noise, the ability of the resident to maintain its territory was reduced.

Simpson et al. (2016) found that motorboat noise elevated the metabolic rate in Ambon damselfish *(Pomacentrus amboinensis*), which when stressed responded less often and less rapidly to simulated predatory strikes from their predator, the dusky dottyback (*Pseudochromis fuscus*). Damselfish were captured more readily by their predator during exposure to motorboat noise compared with ambient conditions, and more than twice as many damselfish were consumed by the predator in field experiments when motorboats were passing. This study suggests that boat noise in the marine environment could impact fish populations and predator-prey interactions.

There is considerable concern about the effects of noise on cetaceans. Nowacek et al. (2007) reviewed responses of cetaceans to noise and found three types of responses: behavioral, acoustic and physiological. Behavioral responses involve changes in surfacing, diving, and swimming patterns. Acoustic responses include changes in the timing or type of vocalizations. Physiological responses include shifts in auditory thresholds. In this study they documented responses of cetaceans to various noise sources but were concerned about the absence of knowledge about effects of noise sources such as commercial sonars, depth finders and acoustics gear used by fisheries. Romano et al. (2004) measured blood parameters of the beluga whale, *Delphinapterus leucas,* and bottlenose dolphin, *Tursiops truncatus,* exposed to noise. Norepinephrine, epinephrine, and dopamine levels, neurotransmitters related to stress, increased with increasing sound levels, and were significantly higher after high-level sound exposures compared with low-level sound exposures or controls.

Noise from ship traffic and commercial, research and military activities has increased greatly over the past century and has caused changes in the vocalizations and behaviors of many marine mammals, including beluga whales (*Delphinapterus leucas*) (Lesage et al.,1999), manatees (*Trichechus manatus*) and right whales (*Eubalaena glacialis*, *E. australis*) (Parks et al., 2007). The calls of killer whales are prolonged in the presence of noise from boats, probably to compensate for the acoustic pollution (Foote et al., 2004), while humpback whales (*Megaptera novaeangliae*) increase the repetition of phrases in their songs when exposed to low-frequency sonar (Miller et al., 2000). Several species of dolphins change their behavior and vocalizations when there is boat noise (Buckstaff, 2004). Parks et al. (2011) documented calling behavior by individual North Atlantic right whales (*Eubalaena glacialis*) in the presence of increased background noise. Right whales respond to periods of increased noise by increasing the amplitude of their calls, which may help to maintain their communication range with conspecifics during periods of increased noise. This may be interpreted as an adaptive response. However, periods of high noise are increasing and have reduced the ability of right whales to communicate with each other by about two-thirds. E. glacialis were studied by Hatch et al. (2012) in an ecologically relevant area (the 10,000 km^2 Stellwagen Bank marine sanctuary) and time period (one month) using vessel-tracking data from the U.S. Coast Guard's Automatic Identification System to quantify acoustic signatures

of large commercial vessels and evaluate the noise from vessels inside and outside the sanctuary. By comparing noise levels from commercial ships today with the lower noise conditions a half-century ago, the authors concluded that right whales have lost about 63 -67% of their communication space in the sanctuary and surrounding waters.

4.4 Loss of Species/Impacts To Ecosystem Services

As noted above, the scientific literature is in general agreement that while there are important commercial and economic benefits to dredging, there are also environmental impacts associated with dredging operations that need to be addressed. The following sections outline the steps that can be taken to minimize those impacts.

4.5 Habitat disruption

In addition to the direct removal of sediment and benthic biota, and noise impacts described above, dredging activities may result in bank erosion, change in bottom substrate characteristics and channel flow characteristics within dredged waterways.

Wu et al. (2017) modeled effects of dredging (light attenuation from sediment disturbance) to seagrasses of the genera *Amphibolis* (Australia), *Halophila* (Australia) and *Zostera* (Australia). The genus *Zostera* is found along the East Coast of North America. Although impacts varied with dredging parameters and the seagrass meadow being studied, in general, dredging over three months of duration or more, or repeat dredging every three or more years, were key thresholds beyond which resilience of seagrass beds could be compromised. The authors also concluded that managing light reduction to less than 50% could decrease seagrass loss, recovery time, or risk of local extinction.

5. MINIMIZING DREDGING IMPACTS

As noted above, the scientific literature is in uniform agreement that while there are uniform benefits to dredging, there are many serious environmental impacts that are associated with those dredging activities. The following sections outline the steps that are taken to minimize those impacts.

5.1 Dredging Environmental Windows for Species of Concern

Timing restrictions are employed to reduce the risk of impact by project activities to sensitive life stages of some federal- and state-managed species and habitats. Impacts from dredging include entrainment, sedimentation, potential contamination, disturbance from sound impacts, and turbidity impacts. Recommendations for timing restrictions provided by resource agencies are important for the conservation of state and federal resources, however the resultant windows of time available for work often limit the time available for dredging projects to be accomplished.

Federal restrictions on dredging are based on the Endangered Species Act (ESA), Fish and Wildlife Coordination Act (FWCA), and Magnuson-Stevens Act (MSA) authorities. State restrictions are provided within the NJDEP. State and federal timing windows and work restrictions are often similar, and there is often coordination between the state and federal agencies issuing the

recommendations. The recommendations are provided by NJDEP for the state permit; the federal agencies provide recommendations to the lead federal action agency, typically the USACE.

Timing restrictions that are especially impactful for dredging operations in New Jersey include those for winter flounder, anadromous fish species which are at or below management thresholds {Conventional (agreed values) of indicators of the desirable or undesirable state of a fishery resource}, such as river herring and shad, and endangered Atlantic and shortnose sturgeons, and threatened piping plover. Other restrictions may be provided for dredging activities in New Jersey when dredging may impact non-EFH entities such as marine mammals, sea turtles, shellfish beds, SAV, blue crabs, oyster reefs, horseshoe crabs, seabeach amaranth, and nursery grounds for several species of sharks.

For such species as river herring, the sturgeons and piping plover, timing restrictions are location-based; if a dredge project is planned for an area where sensitive stages of those species are known to occur (e.g., in spawning or nursery areas) or where there is mapped critical habitat, a timing restriction may be provided during the permitting process. NMFS provides an evaluation tool for the lead federal action agency to complete with project specifics (e.g., duration of activity, sound impacts, and vessel traffic impacts, etc.) to determine whether the project is likely to impact sturgeon and other endangered species. However, the evaluation tool is not available for non-endangered NOAA resources, e.g., river herring.

From the Federal perspective, dredging windows established pursuant to the ESA, the FWCA, and the MSA will be in effect on a nationwide basis, depending upon the distribution of the particular species of concern. However, like New Jersey, many other coastal states have their own environmental windows for species or habitats of concern that have to be addressed when planning for a dredging project. While due to the age of the document a dredging operator or project proponent would need to verify the current status of a particular window in a particular state, but Lukens (2000), provides an excellent summary of the status of dredging and dredged material management programs for U.S. coastal states, territories, and commonwealths, including states in the Great Lakes region.

Potential for Changes to Environmental Windows

For managed species such as winter flounder, timing restrictions are based on guidelines established from habitat characteristics and life history parameters within the range of the species. For federally managed species, the guidance is based on EFH source documents, as mandated by the MSA, which are compiled by NMFS and managed by the NEFMC and MAFMC. Typically a timing restriction is recommended based on the project's location within the species' habitat and if the project activity will impact the habitat, migration pathways, or a sensitive life stage of the species.

As part of the process to permit a dredging project, the applicant, whether a Federal agency such as the U.S. Corps of Engineers, or a private entity such as a local marina, is required to provide information to the regulatory agencies regarding the hydrologic, bathymetric, geotechnical, and environmental aspects of the area to be dredged. It is during this time that windows that limit the execution of a project are identified, and ultimately included as condition of any issued Federal or state permit for the dredging project. For both federal and state applications, the applicant may

appeal to the agency issuing the timing restriction for an exemption to the timing restriction or permission to extend work. In considering the appeal, the resource agency may consider specific details of the project activity (e.g., duration, sequencing of multiple activities), sound impacts, turbidity impacts, as well as environmental characteristics of the project location.

At the state level, the State of New Jersey ultimately has the ability to make policy and regulatory decisions regarding how and why dredging windows are identified and enforced. However, at the Federal level, the dredging window for various species are predicated on regulations included within the MSA, and based on technical decisions prepared by either the NEFMC or the MAFMC. Actual changes to the dredging windows will not occur without extraordinary research efforts to demonstrate the habitat requirements or life history elements of the species have changed. For example, in October 2017, the NEFMC issued an amendment to the EFH source document for winter flounder (see https://www.nefmc.org/library/omnibus-habitat-amendment-2), modifying some EFH elements, including parts of the environmental windows.

That being said, there are limited case-by-case situations where NMFS has allowed for some slight easement of the dredging windows. This has usually consisted of allowing the project to extend a few days into the window or potential start a few days early. But the NMFS typically does not exclude a project from a dredging window as that would be counter to the MSA. Some slight exceptions have been granted, but only based on detailed discussions with NMFS. Such discussions do require the applicant to have significantly more information regarding the environmental/biological characteristics of the area being dredged then would normally be required in a standard dredging permit application.

Winter Flounder Case Study for Federal Permit Evaluation

To more fully understand how a permit is evaluated with respect to dredging windows, a hypothetical application with winter flounder as the species of concern follows. Winter flounder is a federally managed species for which a very broad timing restriction is issued throughout New England and the Mid-Atlantic. In New Jersey, a timing restriction of January 1 through May 31 is recommended for projects from northern New Jersey to Absecon Inlet. Winter flounder eggs are adhesive and stick in clumps to the bottom of a variety of substrates in Mid-Atlantic and New England estuaries. Larvae are associated with the bottom during certain stages in their development. Winter flounder adults move into estuaries in the late fall; spawning begins when water temperatures decrease to 10C (approximately 50F). In the Mid-Atlantic, this generally occurs in mid to late December. The timing restriction is recommended to protect eggs and developing larvae; the timing restriction recommendation was re-authorized in January 2018 by the New England Fisheries Management Council under the authority of the Magnuson-Stevens Act (NEFMC 2018).

Impacts are evaluated on an EFH assessment worksheet (https://www.greateratlantic.fisheries.noaa.gov/habitat/efh/efhassessment.html) which is completed by the applicant or the USACE project manager; the assessment allows the applicant to provide detailed information regarding environmental parameters at the project site. An application receives a full review from NMFS for impacts to EFH for federally managed species and their prey. For the winter flounder component of the review, special attention is made to EFH for spawning adults, eggs, and larvae, since the early life stages are especially vulnerable to certain

impacts, such as dredging. Parameters at the site location and impacts from the project are compared to information on winter flounder provided by NEFMC and NEFSC, as regulated by MSA authorization. Timing restrictions are provided to the USACE in the form of conservation recommendations; these recommendations are evaluated by the USACE in their permit review; the USACE makes the final decision about whether or not to incorporate the timing restrictions (as special conditions) into the permit.

Occasionally an applicant will request a modification to the timing restriction. This typically occurs if an applicant is actively engaged in dredging before implementation of the timing restriction and determines that dredging will not be completed before the timing restriction begins. NMFS will typically look at prevailing water temperatures in the project area (based on real-time data from federal or state water monitoring stations). If water temperatures are above 10° C (winter flounder spawning occurs at 10°C and below), then a few days incursion into the timing restriction is recommended.

5.2 Site Characterization

Dredging operations have specific impacts, such as entrainment of benthic eggs and larvae and generation of sound and turbidity, which are unavoidable regardless of the type of dredge employed in the operation. Species-specific limiting parameters, such as the temperature at which winter flounder begin to spawn, are available in EFH source documents for all federally managed species on the NEFMC and MAFMC websites. It therefore may be possible to characterize a site to determine if a project location could be considered exempt from a timing restriction based on whether or not the physical characteristics of the site are appropriate for the species in question. However, a single site assessment may not be appropriate for such evaluations; the use of long-term data sets on such parameters as temperature, salinity, current velocity and sedimentation rate would enable the resource agencies to better evaluate site-specific characteristics. As federal agencies such as NOAA and USFWS, in addition to NJ state agencies, are involved in these decisions, it is incumbent upon both state and federal resource agencies to establish site evaluation guidelines to enable the permitting agencies to appropriately characterize a project site. A dredging workshop held in New York in 2011 also recommended increased communication among state and federal agencies in order to better evaluate dredging impacts (Tanski et al., 2014).

The current regulatory approach to approving a dredging application is the evaluation of submitted information regarding the physical, chemical and biological characteristics of the proposed dredge site. A hydrographic survey of the dredging site has to be taken within 6 months prior to submission of the application. The regulations require an inventory of aquatic resources in the area to be dredged such as shellfish beds, eel grass beds, wetlands, shorebird nesting habitat, migratory pathways and/or spawning habitat for finfish, and other aquatic organisms. Sediment characterization data such as grain size, total organic carbon, percentage moisture, and bulk sediment chemistry analysis data also need to be provided.

5.3 Advances in Dredging Technology

While there are recognized concerns with dredging, because of the economic need of dredging as a tool for improving navigation in our nation's waterways, both the dredging industry and the federal government are continually working to improve both the tools used during dredging

operations and the management of those tools. Much of that work comes through the USACE via programs such as the Dredging Operations and Environmental Research (DOER) program. In general, the focus of the program is in improving how dredging is conducted, as well as improving the technology by which dredging is accomplished.

The improvement in dredging management is in the development of computer hardware and software programs such as Silent Inspector, which is an automated system used to help control the accuracy of dredging depth in certain types of dredges, as well as help the dredge operator comply with permit limitations such as depth of dredging, location of dredging or other such item that may be placed as a result of T&E species considerations. Other software improvements include increased ability to monitor turbidity and verify that over-dredging is not occurring.

From a dredging technology standpoint, research is focused on efforts to improve both the efficiency of dredging operations, as well as attempts to limit the broader impacts of dredging. There are any number of improvements in dredging technology that have helped refine the ability to efficiently dredge, including modified and improved cutterheads, improvements in hydraulic pumps, and the use of water injection technology for dredging purposes. For environmental dredging, that includes improvements in the use of closed buckets to control suspended sediment (Bridges and Russo 2007, and Russo et al. 2012). Additional improvements include the development of smaller pieces of dredging equipment (so-called mini-dredges), that have the ability to work in more confined areas and, because of their physical size, are less likely to impact the environment (DOER 2000).

6. UNCERTAINTY/LIMITATIONS

Dredging of accumulated sediment from freshwater or marine water bodies for navigation channel deepening and recreational purposes can reduce short, and long-term risks to navigation and better accommodate deep-draft commercial ships. Such removal can be highly effective for accommodating the movement of commerce but much less effective for reduction of overall environmental risk due to resuspension of sediments during dredging operations (see section 4. Impacts of Dredging). For example, a frequently cited concern regarding the potential detrimental impacts to fish, benthic and sessile organisms associated with dredging projects is sediment resuspension resulting from the excavation process, overflow, or open-water placement (Clarke and Wilber, 2000; Clarke et al. 2015). Historically, dredging has been managed to prevent crossing a critical threshold value (Savioli et al. 2013). While this static monitoring approach is common in the dredging industry because it establishes a pre-defined environmental constraint, it has broad limitations because it does not address the spatial and temporal mechanisms by which sediment plumes move through the environment (the primary concern of many regulatory agencies). Inherent in these limitations are the uncertainties associated with measuring the effects of sediment resuspension on aquatic fauna because uncertainty is an intrinsic factor in all data collection, estimation techniques, and simulations used in sediment quality assessment. Although estimates of total suspended solids (TSS) or turbidity are often stated as specific values, there is always a degree of uncertainty associated with the estimate and that uncertainty must be understood and communicated to those making resource management decisions.

Uncertainty arises from the inability of our data-collection and analytical procedures to fully characterize the natural spatial and temporal variability associated with environmental unpredictability (e.g., changes in hydrology, geology, climate, and land use). Uncertainty is also present in models because it is impossible to reproduce a natural system in a model with complete accuracy. There are generally two types of uncertainty associated with scientific assessments – observational uncertainty and simulation uncertainty. Observational uncertainty is error associated with the measurement of a target attribute such as turbidity or TSS concentrations whereas simulation uncertainty is the error associated with estimating the concentration of TSS for time periods or for locations where measurement is not possible. Both of these sources of uncertainty need to be addressed as part of any dredging process to reduce under-estimating exposure risk. A report by the US Environmental Protection Agency provides an overview of case studies of environmental dredging projects (USEPA 2003) and discusses many of the risks and limitations associated with dredging – these will not be reiterated here. However, it is germane to this report and to managers and decision makers to develop monitoring strategies that incorporate a better understanding of the uncertainty associated with dredging operations.

For example, climatological changes may compound this uncertainty and resource managers may also need to consider how these changes affect the life history of species of concern. It is known that many species of fish are migrating further north toward colder waters as our waters are warming. Nye et al. (2009) analyzed temporal trends from 1968 to 2007 in the mean center of biomass, mean depth, mean temperature of occurrence, and area occupied in 36 fish stocks in the northeast Atlantic of the US. Temporal trends in distribution were compared to time series of both local- and large-scale environmental variables, as well as estimates of survey abundance. Many stocks comprising different taxonomic groups, life-history strategies, and rates of fishing showed a poleward shift in their center of biomass, most with a simultaneous increase in depth, and some with a concomitant expansion of their northern range. Stocks located in the southern extent of the survey area exhibited much greater poleward shifts in center of biomass and some occupied habitats at increasingly greater depths.

In addition, climate change can also modify the timing of migration, reproduction and egg deposition. There have been extensive studies on the phenology (timing) of terrestrial species, but fewer studies on how climate variability affects seasonal behavior of marine species. Warming ocean temperatures may also affect migration patterns. Asch (2015) studied the larval stages of 43 fish species collected off the Southern California coast between 1951 and 2008. She found that the phenology (breeding) changed over the years; 39% of breeding events occurred earlier in the season in recent decades. In light of these ongoing changes, it will be important to verify that the species we are trying to protect with dredging windows (see section 5.1 Dredging Windows for Species of Concern) are indeed using established migration pathways or continue to inhabit historic nursery grounds at the time expected. Adjusting the timing of seasonal management tactics (such as dredging windows) can help ensure that management remains effective.

While not often considered in terms of smaller or more focused dredging projects, cost benefit analysis or net environmental benefit analysis may help capture some of that uncertainty. For large dredging projects such as the Delaware River Main Channel Deepening Project initiated in 2002, a comprehensive economic analysis has to be conducted as part of the NEPA Environmental Impact Statement (EIS) process. However, for most dredging projects of smaller magnitude, a

NEPA Environmental Analysis (EA) or EIS is not required as part of the permitting process through the USACE or the NJDEP. As such, consideration of the cost implications of a small dredging project may help with the justifications of how to approach a project with tangible environmental benefits, but extreme timing constraints from the perspective of scheduling. However, it is noted that the removal or waiving of a Federal timing window is not possible solely on a cost-benefit analysis. But it may assist with discussions with the regulatory agencies relative to the length or the implementation of the timing windows.

Adaptive management strategies can be specifically aimed at addressing the non-stationarity of environmental processes (Milly et al. 2015) – that is, moving away from a static one-time assessment to one that incorporates uncertainty associated with changes in species occurrence and distribution and integrates the spatial and especially the temporal complexities of monitoring sediment plumes. Savioli et al. (2013) lay out four elements that are paramount to implementing an adaptive management strategy: 1) The collection of baseline information prior to the dredging project implementation phase; 2) Field monitoring that includes measurement and simulation modelling procedures; 3) Evaluation of data and results; and 4) Adaptation. Adaptation includes not only the re-assessment of the implemented dredging strategy but also the evaluation of the objective target values that are usually quite uncertain. In this way it is possible to adapt the dredging procedures to the conditions at the site thus minimizing impacts to sensitive life stages and species of concern (e.g., Berry et al. 2011, Lackey et al. 2009) while optimizing dredging operations. This adaptive management approach can not only be used to better understand spatiotemporal changes but also allows managers to develop uncertainty bounds around the target values (i.e., observational uncertainty) especially if turbidity and TSS levels fluctuate both during and after excavation.

Adaptive management can be implemented as a proactive process designed to minimize the impacts from dredging around sensitive areas. The proactive approach is typically defined in an environmental feedback monitoring and management plan (Savioli et al. 2013) and incorporates control of potential impacts, instrumentation to assess the potential health of the sensitive receptors (see also Clarke et al. 2007), predictive modeling that can be used to provide a detailed temporal and spatial picture of potential impacts, and assessment of as sensitive receptors reaction to impacts (i.e., a feedback loop). Close coordination between the dredging contractor and the environmental monitoring team needs to take place in order to identify the best approaches and minimize impacts to sensitive receptors. This type of feedback loop is particularly important to insure that mitigation measures are taken into consideration during and after dredging operations. Additionally, simulation modelling can be extremely important in analysis of special and temporal changes because it provides a link between dredging operations and the monitoring data. As part of an adaptive management approach, compliance monitoring and reporting need to be carried out to confirm that dredging operations are meeting the sediment quality objectives. Often, it is the compliance monitoring that falls by the wayside because budget limitations often restrict follow-up assessment. Ultimately, the implementation of an adaptive environmental monitoring program can be highly beneficial during and after dredging as it can help optimize dredging operations and also provide entities like the NJDEP with a mechanism for assessing the uncertainty associated with meeting sediment quality guidelines and may help prevent future impacts on vulnerable life stages of fish and benthic organisms.

7. RECOMMENDATIONS

The literature to date has focused mostly on effects of suspended sediments. There is a need for more study on the other types of stresses caused by dredging, such as noise, contaminated sediments, and entrainment. There is a great need for more field studies (*in situ*). The combination of toxicity, chronic stress, loss of habitat, and repeated exposure to multiple contaminants and pulses of suspended sediments can result in synergistic effects. Since eggs and larvae are most vulnerable, dredging windows should be used to avoid stressing these early life stages.

Indirect effects, such as loss of prey and habitat changes, are also in need of study. A BACI design (Before-After-Control-Impact) is appropriate for field studies to assess multiple responses of targeted life history stages to multiple stresses over time.

The EPSC would note that the overriding restriction on regulatory approval to dredging projects are the timing restrictions implemented by the federal government. NMFS's mandates come from ESA and from MSA and FWCA authority. MSA authority is based on guidance from NEFMC and MAFMC. That being said, modifications of timing restrictions were made previously in NY Harbor based on 10 years' worth of data provided by NY ACOE on the Harbor Deepening Project. This is an example of why long-term data sets characterizing a site (rather than a short-term snapshot) are important in order for any changes to be made to timing restrictions.

It is also worth noting the recommendations from Tanski et al. (2014). This paper note that dredging windows are intended to be a tool to protect living resources from unacceptable damage, but generic fisheries windows based on risk avoidance are no longer workable because they restrict dredging time frames to the point where it becomes extremely difficult to perform required dredging. Under an approach based more on risk management, windows could be adjusted if regulators have sufficient information to decide that a project will not cause unacceptable damage. This evaluation has to be done on a project-specific basis. However, such an approach would require some level of regulatory relief at the Federal level to allow for such approaches to work.

Specific recommendations:

- As permit limitations are based primarily on conservative dredging time of year restrictions (windows), which are often not fully based on the current condition of the species, consider research that can be useful in refining windows or in reducing exposure time of sensitive live stages.
- Explore collaborative funding with federal resource agencies for the purposes of more basic science on specific dredging impacts within New Jersey waterways. This may include basic inventory studies to supplement our existing knowledge of species usage within different water bodies of the state (e.g. Delaware River nursery areas for striped bass). Accumulating this information may eventually provide the basis for GIS databases that will facilitate decisions regarding where and when to allow dredging.
- Establish a workgroup between relevant NJDEP departments and federal regulatory agencies such as USACE (Philadelphia and New York districts), USFWS and NMFS to establish guidelines for applicants to use to request modifications or exemptions to timing restrictions.

- Consider establishing a workgroup between relevant NJDEP departments and other states in the region (New York, Connecticut, Delaware, Maryland, and Pennsylvania) to examine regional limitations in dredging windows and assess approaches to streamlining state permitting by developing pertinent data useful in managing the windows.
- Based upon results of inter-agency coordination and establishment of guidelines for applicants, compile existing life history data by species within different geographic areas of the State that may be subject to dredging projects. Construct a database for use and reference by both the NJDEP and regulated public that would provide a basis for impacts assessment of dredging projects.
- Determine locations where long-term environmental data sets would be needed in the evaluation of permit modifications or exemptions, and provide state funding for establishing long-term monitoring and adaptive management strategies that incorporate uncertainty associated with changes in species occurrence and distribution and integrate the spatial and temporal complexities of sediment plumes.
- Determine locations where long-term species distribution and range data would be needed in the evaluation of permit modifications or exemptions, and provide state funding for surveys.

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Appendix A

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ORIGINAL ARTICLE

WILEY FISH and FISHERIES

A critical analysis of the direct effects of dredging on fish

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Abstract

Dredging can have significant impacts on aquatic environments, but the direct effects on fish have not been critically evaluated. Here, a meta-analysis following a conservative approach is used to understand how dredging-related stressors, including suspended sediment, contaminated sediment, hydraulic entrainment and underwater noise, directly influence the effect size and the response elicited in fish across all aquatic ecosystems and all life-history stages. This is followed by an in-depth review summarizing the effects of each dredging-related stressor on fish. Across all dredging-related stressors, studies that reported fish mortality had significantly higher effect sizes than those that describe physiological responses, although indicators of dredge impacts should endeavour to detect effects before excessive mortality occurs. Studies examining the effects of contaminated sediment also had significantly higher effect sizes than studies on clean sediment alone or noise, suggesting additive or synergistic impacts from dredging-related stressors. The early life stages such as eggs and larvae were most likely to suffer lethal impacts, while behavioural effects were more likely to occur in adult catadromous fishes. Both suspended sediment concentration and duration of exposure greatly influenced the type of fish response observed, with both higher concentrations and longer exposure durations associated with fish mortality. The review highlights the need for *in situ* studies on the effects of dredging on fish which consider the interactive effects of multiple dredging-related stressors and their impact on sensitive species of ecological and fisheries value. This information will improve the management of dredging projects and ultimately minimize their impacts on fish.

KEYWORDS

contaminated sediment, dredging impacts, fisheries, meta-analysis, noise pollution, suspended sediment

1 | INTRODUCTION

Dredging involves the excavation and relocation of sediment from lakes, rivers, estuaries or sea beds and is a critical component of most major marine infrastructure developments along the coast (dredging, the fishing technique commonly associated with the catch of bivalves, is not discussed in this review; but see Reine, Dickerson, & Clarke, 1998; Watson, Revenga, & Kura, 2006). The removal of seabed sediments is commonly used to create or maintain navigable depths for shipping channels and harbours and provide material for land reclamation and coastal development projects. Material may also be dredged for the purpose of beach replenishment and mineral and/or gas extraction from underwater deposits (USACE 1983). The expansion of port facilities to accommodate the new generation of large-capacity

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vessels, and continued development of offshore energy resources will also require an increase in dredging services.

Globally, dredging methods include both mechanical (e.g. grab and excavator dredges) and hydraulic (e.g. trailer suction hopper and pipeline cutterhead dredges) processes (USACE 1983; VBKO 2003). Dredging in coastal marine waters generally requires hydraulic dredges to obtain economic efficiencies for sustaining high production rates. Dredging often has two main sites of operations, the dredge site and the dredged material disposal site. In addition to direct impacts at these sites, sediment plumes can extend several kilometres from the dredging operations, depending on the quantities and grain-size composition of the dredged material and local hydrodynamic conditions (Evans *et al.*, 2012; Fisher, Stark, Ridd, & Jones, 2015). Local physical and environmental conditions, as well as the scale and method of dredging, determine the spatial and temporal scale of the exposure that aquatic organisms experience during dredging-induced perturbations (Bridges *et al.*, 2008; PIANC 2009; Wilber & Clarke, 2001). Scales and modes of impact are also dependent on whether the project involves capital dredging (excavation of previously undisturbed sediment) or maintenance dredging (periodic removal of accumulated sediments following construction) and the history of the site that is to be dredged. A distinction must also be made between scales of impact associated with excavation vs. placement processes. A detailed characterization of diverse dredging methods and their sediment release mechanisms is beyond the scope of this study, but it is recognized that knowledge of dredging processes is a prerequisite for an accurate risk assessment of a dredging project.

Despite the necessity of dredging for industrial development, its potential impacts on the environment are of particular concern as multiple potential stressors associated with dredging activities have been well documented. Chief among these are sediment stress (suspended and deposited), release of toxic contaminants, hydraulic entrainment and noise pollution (Figure 1; McCook *et al.*, 2015; Reine & Clarke, 1998; Reine, Clarke, & Dickerson, 2014; Reine, Clarke, Dickerson, & Wikel, 2014; Wilber & Clarke, 2001). Although there are significant dredging operations undertaken across a range of aquatic environments, and an increasing body of literature documenting dredging-related effects on fish is available (e.g. Wenger *et al.* 2015), our knowledge of the relationships between multiple dredging-related pressures and of their cumulative or interactive effects on fish is still poor. Fish are ecologically, economically and culturally important components of all aquatic environments, with millions of people relying on fish for food or income, thus warranting further investigation into how they are impacted by dredging. Reviews on the effects of dredging-related stressors on fish have previously focused on solitary stressors, such as exposure to elevated suspended sediment concentrations (e.g. Kerr, 1995; Newcombe & Jensen, 1996; Wilber & Clarke, 2001). Effects from multiple dredging components on fish, however, have yet to be synthesized. Such knowledge is critical for predicting potential impacts and designing appropriate, fish-focused management

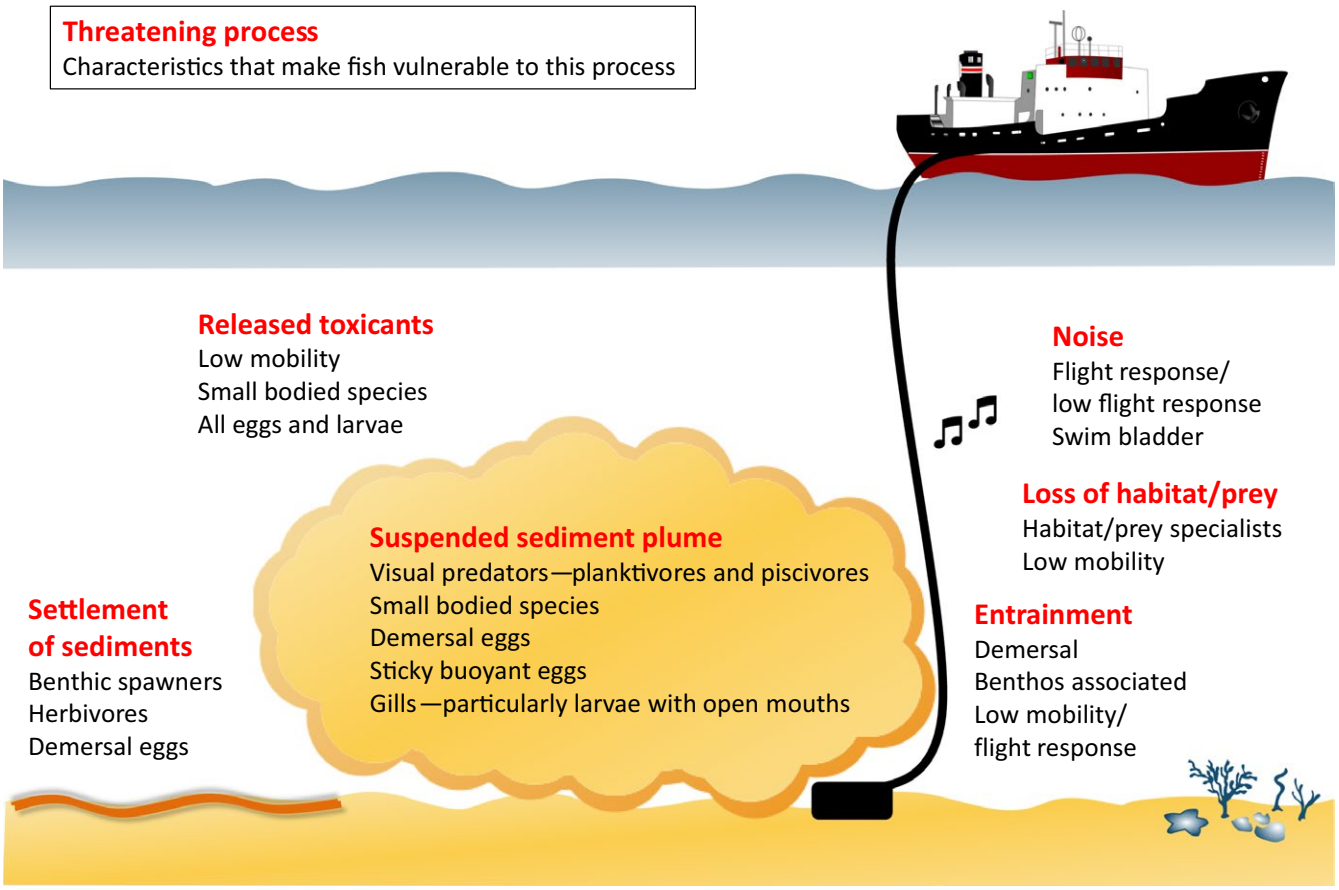

FIGURE 1 A schematic diagram of categories of potential effects of dredging on fish. [Colour figure can be viewed at wileyonlinelibrary.com]

strategies, which avoid or minimize potential impacts, but do not unnecessarily constrain dredging activities (Kemp, Sear, Collins, Naden, & Jones, 2011; NAS, 2001; PIANC 2009). Consequently, reviews of the state of knowledge of dredging-induced impacts and identification of knowledge gaps are an essential first step in determining effective risk reduction measures, and developing best management practices (NAS, 2001; PIANC 2009).

Ultimately, the risk of detrimental impacts depends on exposure characteristics, in particular intensity and duration, and on the tolerance thresholds to the various stressors for the fish species of concern (ANZECC and ARMCANZ 2000; Browne, Tay, & Todd, 2015; Erftemeijer & Lewis, 2006; Wilber & Clarke, 2001). If both the exposures and responses are accurately assessed, appropriate risk management measures can be identified to balance the need to construct and maintain coastal infrastructure with adequate protection of vulnerable species and valuable finfish fishery resources. This review and meta-analysis synthesizes and characterizes the known direct effects on fish from exposures to the most commonly cited potential stressors associated with dredging: sediment, release of toxic contaminants, hydraulic entrainment and noise (McCook *et al.*, 2015; Reine & Clarke, 1998; Reine, Clarke, & Dickerson, 2014; Reine, Clarke, & Dickerson, 2014; Reine, Clarke, Dickerson, & Wikel, 2014; Wilber & Clarke, 2001), with an emphasis on exposures relevant to dredging processes.

2 | METHODS

2.1 | Development of framework for the review

The development of this review was undertaken at a workshop in October 2013 by stakeholders from state and federal government agencies, including the Environment Protection Authority (Western Australia), Western Australia Department of Fisheries and Department of Parks and Wildlife, and the Australian Institute of Marine Science; experts from multiple universities; and representatives from private industry. The overall objective of the workshop and the assessment was to synthesize and quantify the effects of dredging-related pressures on critical ecological and physiological processes for finfish and critically evaluate the factors that influence the effects of dredging on fish. To identify what the potential impacts of dredging could be, previous studies and reviews on the effects of dredging on aquatic organisms were assessed as a group. Literature on impacts of dredging was found through Google Scholar, Scopus and the ISI Web of Knowledge, using the search terms "dredg*," "impact*," "effect*" and "environment*." Results that did not pertain to dredging as defined in our review were filtered out. Results that did not mention particular impacts or environmental responses associated with dredging were also excluded. For the purposes of creating an initial list of impacts, all potential impacts were recorded, regardless of the aquatic organism it was shown to affect. Articles were also provided by stakeholders with particular dredging expertise. In the end, 33 sources of information were used to compile a list of environmental impacts associated with dredging (Table S1).

There were six main potential impacts identified as associated with dredging: habitat loss, hydraulic entrainment, release of contaminants, sedimentation, suspended sediment and underwater noise (Figure 1). The strong relationship between fish and habitat means that any direct impact on habitat will affect most fish species (e.g. Jones, McCormick, Srinivasan, & Eagle, 2004). Habitat loss and degradation can be a major aspect of the impact of dredging on fish communities (Amesbury, 1981; Galzin, 1981; Lindeman & Snyder, 1999). Dredging-induced habitat loss was considered to have an indirect effect on fish, and as this has been reviewed previously (e.g. Erftemeijer & Lewis, 2006; Erftemeijer, Riegl, Hoeksema, & Todd, 2012) and is generally already considered during the approval process for proposed dredging works (Erftemeijer *et al.*, 2013; PIANC 2009), it was not considered in this review. Ultimately, the overarching objective for this review was to characterize the direct effects of dredging impacts on fish. The protocol used to search the literature is described below.

2.2 | Review protocol

Literature was sourced from Google Scholar, Scopus and the ISI Web of Knowledge using search terms relevant to each potential impact. The following search terms were used: ["suspended sediment*" OR "sedimentation" OR "turbid*" OR "dredg*"] AND "fish*"; "suspended sediment*"AND ["contam*" OR "metal*" OR "PAH*" OR "PCB*" OR "OCP*" OR "organochlor*"] AND "fish*"; "dredg*" AND "entrain*" AND "fish*"; "Dredg*" AND "sound" AND fish"; "Dredg*" AND "noise" AND "fish"; "Contin*" AND "sound" AND "fish"; "Contin*" AND "noise" AND "fish"; "Noise" AND "fish"; "Sound" AND "Fish." Relevant articles from reference lists of papers were used to identify additional sources of literature. In addition, unpublished grey literature, reports and management plans were identified and sourced through consultation with the stakeholders present at the workshop.

Beyond being relevant to each impact, to be included, studies needed to state the fish species and life-history stage being tested, have a clear experimental design (i.e. could be repeated), state concentrations and exposure times used (when experimental), have a clear experimental endpoint and present data in units that could be compared to other studies. To be conservative, the data that were extracted from each study were the lowest concentration where a specific effect was observed. If no effect was observed, the highest concentration that did not elicit an effect was extracted.

2.3 | Meta-analysis

Once the results of each study were extracted, they were ranked by type of response, which facilitated comparison across stressors (Table 1; see ranks of each study in Tables S2–S5). Where possible, the Hedges' *g* effect size (absolute value) of each study was calculated (Equation 1; Tables S2–S5).

$$g = \frac{\overline{X_1} - \overline{X_2}}{\sqrt{\frac{(n_1-1)S_1^2+(n_2-1)S_2^2}{n_1+n_2-2}}} \quad (1)$$

TABLE 1 The types of effect ranked to facilitate comparison

Rank	Type of effect
0	No effect
1	Minor behavioural changes—avoidance of a stressor
2	Minor physical damage—gill damage, skin abrasions and changes to development times, OR Moderate behavioural changes—reduced foraging rate or changes to habitat association, but did not record any physiological changes
3	Physiological changes—changes in hormone levels, reduced growth rate, organ function or developmental abnormalities
4	Increase in mortality or reduced hatching success

where $\overline{X_1}$ equals the mean of the treatment group response, $\overline{X_2}$ equals the mean of the control group response, n_1 is the sample size of the treatment group, n_2 is the sample size of the control group, and S_1 and S_2 are the standard deviations of the treatment and control groups, respectively. We chose Hedges' *g*, as it is more robust for studies with small sample sizes (Hedges, 1981). To examine potential drivers of variability in effect sizes across all stressors, we generated a generalized mixed-effects model with the package lme4 in the R programming language (R Development Core Team 2014) using a Laplace approximation and a log link function to meet the assumptions of the model (Bates *et al.* 2014). We evaluated the appropriateness of the model by examining Q–Q normality plots of effect sizes using the package car (Fox & Weisberg, 2011; Figure S1). The models included response type (as described above), habitat (freshwater, estuarine, marine, anadromous, catadromous), stressor type (contaminated sediment, suspended sediment, sound), life-history stage during exposure (eggs, larvae, juveniles, adults), family and the log of exposure duration as fixed effects, and species as a random effect. We performed a linear correspondence analysis (LCA) and calculated the chi-square statistic to examine the association between habitat, life-history stage, or type of stressor and response type using the package Ca in R (Nenadic & Greenacre, 2007).

For each individual stressor, we conducted generalized linear mixed-effects models fit by restricted maximum likelihood to assess potential drivers of effect size. Individual predictors were mean-centred to facilitate model convergence (Wenger, Whinney, Taylor, & Kroon, 2016). To ensure we were meeting the assumptions of the model, we checked the plotted residuals to assess homoscedasticity prior to utilizing the results of the model. We conducted a Wald's test to establish the significance of predictor variables in each model. We further established the robustness of our results by calculating Rosenthal's fail-safe number, an indicator of the number of studies that would need to exist to overturn a significant result (Rosenthal, 1979). A high fail-safe number relative to the number of experiments included in the meta-analysis indicates that the overall effect size of the meta-analysis is a robust estimate of the true effect size (Gurevitch & Hedges, 1999).

For each individual stressor, we also conducted linear discriminant analyses (LDA) using the package MASS in R (Venables and Ripley 2002) to determine the relative influence that the magnitude of the stressor and the exposure time had on the response type. For each LDA, we performed a MANOVA and a Wilks's lambda test to examine whether the explanatory variables had discriminatory power. For each individual stressor, we also performed a linear correspondence analysis and calculated the chi-square statistic to examine the relationship between life-history stage, habitat, source of stressor and response type.

3 | META-ANALYSIS AND REVIEW

Over 430 papers were fully assessed to understand the effects of suspended sediments on fish. Of those papers, the fish response type elicited by suspended sediment was extracted from 59 studies (Table S2). Of those, it was possible to calculate the effect size for 31 data records (Table 2). In addition, 136 peer-reviewed articles were fully assessed to understand the effects of contaminated sediment on fish, from which data records were extracted from 36 articles that directly reported the response type elicited by exposure of fish to contaminated sediment (Table S3). It was possible to calculate the effect size of 25 studies; however, only 12 of these focused on individual contaminants (Table 2; Table S3). Twenty-four publications on the effects of hydraulic entrainment on fish were assessed. From these studies, it was only possible to extract the fish response elicited by hydraulic entrainment from four studies (Table S4). However, it was not possible to calculate the effect size in any of these studies as they all lacked controls. Thirty-five publications were assessed to understand the effects of dredging-related noise on fish. From those publications, we were able to extract the fish response type elicited by sound from 16 studies (Table S5), from which we could calculate effect sizes for nine data records (Table 2).

3.1 | Overall effects of dredging on fish

The results of the generalized linear mixed-effects model indicated effect size is significantly influenced by the type of response observed in fish, the type of stressor and the life-history stage during exposure (Table 3). Studies that recorded increased mortality (response type 4) had significantly greater effect sizes than studies that recorded physiological impacts (Figure 2a). As the objective of many studies that recorded mortality was to find the LC_{50} concentration (the concentration that causes 50% mortality), it is not surprising those that observed mortality had large effect sizes. Hence, this may be an artefact of the type of experiment that produces mortality results and does not necessarily infer that mortality is a good indicator of impacts from dredging. We argue that indicators should detect early signs of stress and allow management intervention before mortality occurs. Studies examining the effects of contaminated sediment also had significantly higher effect sizes than studies on clean sediment alone or noise, suggesting synergistic impacts from dredging-related stressors (Figure 2b).

The results of the linear correspondence analysis and the calculated chi-square statistic reveal there was a significant association

TABLE 2 Derivation of the effect sizes for each study where it was possible to calculate it. Common names and families are all listed in Tables S2–S5 in the Supporting Information

Dredging stressor	Species name	Source	Response (treatment)	Response (control)	Sample size (treatment)	Sample size (control)	*SD* treatment	*SD* control	Effect size (absolute value Hedges' *g*)
Suspended sediment	*Alosa pseudoharengus*	Auld and Schubel (1978)	78.0	84.0	353	353	8.0	9.0	0.70
Suspended sediment	*Alosa sapidissima*	Auld and Schubel (1978)	82.0	95.0	127	127	9.8	4.7	1.69
Suspended sediment	*Clupea harengus*	Johnston and Wildish (1982)	32.7	49.1	8	8	2.7	1.3	7.70
Suspended sediment	*Chromis atripectoralis*	Wenger *et al.* (2013)	70.8	45.8	200	200	66.5	76.4	0.35
Suspended sediment	*Alosa aestivalis*	Auld and Schubel (1978)	71.0	77.0	243	243	18.2	18.9	0.32
Suspended sediment	*Oncorhynchus kisutch*	Galbraith, MacIsaac, Macdonald, and Farrell (2006)	31.2	61.4	4	3	12.6	19.6	1.92
Suspended sediment	*O. kisutch*	Redding, Schreck, and Everest (1987)	11.0	3.0	11	11	19.9	3.3	0.56
Suspended sediment	*O. kisutch*	Servizi and Martens (1992)	1.5	0.2	93	93	3.5	0.5	0.53
Suspended sediment	*Perca fluviatilis*	Ljunggren and Sandström (2007)	0.1	0.2	36	36	0.1	0.5	0.37
Suspended sediment	*Pomacentrus moluccensis*	Wenger and McCormick (2013)	23.3	61.1	20	20	39.8	44.3	0.90
Suspended sediment	*Amphiprion percula*	Hess *et al.* (2015)	37.6	22.4	174	129	54.1	29.5	0.34
Suspended sediment	*A. percula*	Wenger *et al.* (2014)	12.7	11.1	98	91	2.6	1.0	0.78
Suspended sediment	*Clupea pallasii*	Boehlert (1984)	1.7	1.1	5	9	0.3	0.1	2.72
Suspended sediment	*C. pallasii*	Boehlert (1984)	1.5	1.1	5	9	0.2	0.1	2.11
Suspended sediment	*Sander lucioperca*	Ljunggren and Sandström (2007)	0.1	0.1	36	36	0.1	0.1	0.10
Suspended sediment	*Pagrus major*	Isono, Kita, and Setoguma (1998)	40.0	100.0	60	60	38.7	0.0	2.19
Suspended sediment	*Anoplopoma fimbria*	De Robertis *et al.* (2003)	1.5	4.8	6	6	1.7	2.0	1.79
Suspended sediment	*Oncorhynchus nerka*	Galbraith *et al.* (2006)	17.3	41.9	10	10	13.0	17.4	1.60
Suspended sediment	*Acanthochromis polyacanthus*	Wenger *et al.* (2012)	18.1	27.0	25	27	9.5	9.4	0.94
Suspended sediment	*A. polyacanthus*	Wenger *et al.* (2012)	11.5	20.8	25	27	4.5	3.6	2.28
Suspended sediment	*A. polyacanthus*	Wenger *et al.* (2012)	41.2	0.0	3	3	6.1	0.0	9.55
Suspended sediment	*Salmo gairdneri*	Redding *et al.* (1987)	19.0	2.0	11	6	26.5	4.9	0.78
Suspended sediment	*Morone saxatilis*	Auld and Schubel (1978)	68.0	97.0	135	135	18.1	13.3	1.83
Suspended sediment	*M. saxatilis*	Breitburg (1988)	4.9	7.4	19	24	5.2	5.4	0.47
Suspended sediment	*Oplegnathus fasciatus*	Isono *et al.* (1998)	70.0	97.0	60	60	96.8	0.0	0.39
Suspended sediment	*Parapristipoma trilineatum*	Isono *et al.* (1998)	62.0	100.0	60	60	112.3	0.0	0.48
Suspended sediment	*Morone americana*	Auld and Schubel (1978)	49.0	69.0	270	270	34.7	16.1	0.74
Suspended sediment	*Perca flavescens*	Auld and Schubel (1978)	62.0	93.0	165	165	15.9	6.1	2.58
Suspended sediment	*P. flavescens*	Auld and Schubel (1978)	92.0	91.0	333	333	16.0	10.2	0.07
Suspended sediment	*A. sapidissima*	Auld and Schubel (1978)	73.0	80.0	189	189	14.3	29.1	0.31
Contaminated sediment	*Carassius auratus*	Tao, Liu, Dawson, Long, and Xu (2000)	0.9	0.1	4	4	0.1	0.0	11.19
Contaminated sediment	*Hippoglossoides platessoides*	Marcogliese, Nagler, and Cyr (1998)	254.2	154.6	12	15	226.2	220.0	0.45
Contaminated sediment	*Epinephelus coioides*	Wong *et al.* (2013)	17.0	4.0	10	10	8.0	3.0	2.15

(continues)

TABLE 2 (continued)

Dredging stressor	Species name	Source	Response (treatment)	Response (control)	Sample size (treatment)	Sample size (control)	*SD* treatment	*SD* control	Effect size (absolute value Hedges' *g*)
Contaminated sediment	*Oreochromis niloticus*	Peebua, Kruatrachue, Pokethitiyook, and Kosiyachinda (2006)	50.0	0.0	6	6	26.7	0.0	2.65
Contaminated sediment	*Dicentrarchus labrax*	Martins, Santos, Costa, and Costa (2016)	0.2	0.1	10	10	0.0	0.0	3.94
Contaminated sediment	*P. flavescens*	Seelye, Hesselberg, and Mac (1982)	3.1	1.7	10	10	0.2	0.2	6.41
Contaminated sediment	*P. flavescens*	Seelye *et al.* (1982)	2.0	1.5	3	3	0.1	0.0	8.12
Contaminated sediment	*Pleuronectes yokohamae*	Kobayashi, Sakurai, and Suzuki (2010)	22.0	1.0	3	3	2.0	0.0	14.85
Contaminated sediment	*Pimephales promelas*	Sellin, Snow, and Kolok (2010)	2.0	1.3	7	7	0.3	1.1	0.91
Contaminated sediment	*Limanda limanda*	Livingstone *et al.* (1993)	653.3	245.8	5	5	95.7	91.5	4.35
Contaminated sediment	*Oryzias latipes*	Barjhoux *et al.* (2012)	72.1	20.3	3	3	19.4	4.5	3.68
Contaminated sediment	*Leiostomus xanthurus*	Sved, Roberts, and Van Veld (1997)	30.1	33.4	40	40	4.4	4.7	0.71
Contaminated sediment	*Prochilodus lineatus*	Almeida, Meletti, and Martinez (2005)	45.1	23.0	4	6	14.4	4.2	2.35
Contaminated sediment	*Solea senegalensis*	Costa *et al.* (2011)	3.5	1.0	20	20	0.9	0.6	3.31
Contaminated sediment	*Oncorhynchus mykiss*	Brinkmann *et al.* (2015)	9.9	0.7	6	6	3.4	1.2	3.61
Contaminated sediment	*O. mykiss*	Hudjetz *et al.* (2014)	11.6	0.2	10	10	4.3	0.2	3.78
Contaminated sediment	*Scophthalmus maximus*	Hartl *et al.* (2007)	138.0	25.9	8	8	32.0	15.6	4.45
Contaminated sediment	*S. maximus*	Kilemade *et al.* (2009)	135.0	25.9	8	8	27.0	15.6	4.95
Contaminated sediment	*O. mykiss*	Viganò, Arillo, De Flora, and Lazorchak (1995)	1.4	0.3	3	3	0.1	0.1	21.00
Contaminated sediment	*Liza macrolepis*	Chen and Chen (2001)	18.5	0.0	2	2	5.0	0.0	5.29
Contaminated sediment	*O. latipes*	Cachot *et al.* (2007)	44.9	10.0	3	3	16.0	7.0	2.83
Contaminated sediment	*O. latipes*	Vicquelin *et al.* (2011)	42.0	7.8	3	3	4.0	6.7	6.20
Contaminated sediment	*O. latipes*	Vicquelin *et al.* (2011)	88.0	7.8	3	3	8.0	6.7	10.87
Contaminated sediment	*O. latipes*	Vicquelin *et al.* (2011)	68.0	7.8	3	3	2.0	6.7	12.18
Contaminated sediment	*O. mykiss*	Kemble *et al.* (1994)	59.0	0.0	4	4	7.9	0.0	10.62
Sound	*Anguilla anguilla*	Simpson, Purser, and Radford (2015)	0.5	0.4	9	19	0.1	0.1	0.92
Sound	*Multiple species*	Jung and Swearer (2011)	55.0	18.0	8	8	42.4	22.6	1.09
Sound	*C. auratus*	Smith *et al.* (2006)	12.0	39.0	6	6	12.2	12.2	2.20
Sound	*Myxocyprinus asiaticus*	Liu, Wei, Du, Fu, and Chen (2013)	76.1	69.8	5	5	4.9	5.6	1.20
Sound	*C. auratus*	Smith, Kane, and Popper (2004)	165.0	89.7	6	6	44.3	78.4	1.18
Sound	*Cyprinus carpio*	Wysocki, Dittami, and Ladich (2006)	0.4	0.2	6	6	0.0	0.0	3.06
Sound	*Gobio gobio*	Wysocki *et al.* (2006)	0.8	0.4	7	7	1.0	0.1	0.63
Sound	*P. fluviatilis*	Wysocki *et al.* (2006)	0.3	0.2	7	7	0.0	0.0	5.88
Sound	*Sparus aurata*	Celi *et al.* (2016)	163.4	75.6	10	10	117.0	80.5	0.87

TABLE 3 The results of the Wald's test on the generalized linear mixed-effects model examining drivers of effect size overall and within individual stressors

Explanatory variables	Chisq	*df*	Pr(>Chisq)
All stressors			
Response type	20.89	4	<.001
Habitat	1.14	4	.88
Stressor	54.36	2	<.001
Life-history stage	78.1	3	<.001
Log exposure duration	0.53	1	.47
Suspended sediment			
Suspended sediment concentration	0.93	1	.33
Response type	0.24	4	.63
Habitat	2.99	3	.39
Life-history stage	1.29	3	.52
Exposure duration	0.03	1	.86
Contaminated sediment			
Contaminant concentration	1.89	1	.19
Response type	5.26	2	.07
Habitat	4.51	3	.21
Life-history stage	0.84	2	.36
Exposure duration	0.13	1	.72
Sound			
Decibel level	0.97	1	.32
Response type	4.64	2	.03
Habitat	3.7	2	.16
Life-history stage	0.25	2	.61
Exposure duration	0.01	1	.91

between the predictor variables (habitat, life-history stage and type of stressor; $p < 0.01$) and the response type. Visual inspection of the output show studies on larvae and eggs recorded lethal impacts more frequently than other life-history stages. Studies using adult and juvenile fish observed physical damage and physiological impacts most frequently, respectively, while catadromous fishes were most closely associated with behavioural effects (Figure 3). Additionally, the type of responses recorded for fish from freshwater, estuarine and marine environments were very similar, suggesting that results from dredging stressor studies on a range of species can be combined to develop general management guidelines for both marine and freshwater environments.

3.2 | The effects of suspended sediment on fish

A review of studies that have carried out experiments to examine the effects of suspended sediments on fish found the duration of exposure, concentration of suspended sediment, habitat of origin and life-history stages varied considerably among studies. All studies, however, reported continuous exposure lasting between 1.2 min and 64 days across concentrations ranging from 4 to 87,800 mg/L (Table S2). There were 49 records on the effects of suspended sediment on adult fish, 50 records for juvenile fish, 34 records for larvae and 13 for eggs. Forty-nine of the records were from anadromous species, 33 were from estuarine species, 32 were from freshwater species, and 32 were from marine species (Table S2).

There was a wide range of endpoints measured and responses elicited among the studies. Fourteen studies showed no effect of suspended sediment (although only 11 of these recorded an exposure time), 12 studies observed behavioural changes (response type 1), 34 studies recorded physical damage and substantial behavioural changes (response type 2), 37 studies measured physiological stress and sublethal responses (response type 3), and 49 studies recorded some level of mortality (response type 4). Effect sizes ranged from 0.07 to 9.55, with a mean effect size of 1.53 ± 0.33 (*SE*) (Table 2; Table S2).

None of the predictor variables in the linear mixed-effects model significantly influenced variation in effect size of suspended sediments on fish (Table 3). The predictor variables included were suspended sediment concentration, exposure duration, life-history stage and response type. Rosenthal's fail-safe number was 2,870, suggesting that our results are not an artefact of publication bias (Gurevitch & Hedges, 1999). Furthermore, neither sediment type, habitat, nor life-history stage significantly influenced the response type elicited by suspended sediment exposure ($p = .303$) as revealed by the linear correspondence analysis and chi-square test (Table 4).

However, the linear discriminant analysis indicated that increasing both the concentration and exposure time to suspended sediment increased the severity of fish response (Figure 4a,b). Accordingly, the Wilks's lambda results verified the discriminatory power of the explanatory variables ($p < .0001$; Table 4). While there is a clear trend between response type and increasing concentrations and exposure to suspended sediment, fish have markedly different tolerances to suspended sediment, with some species able to withstand concentrations up to 28,000 mg/L, while others experience mortality starting at 25 mg/L (Figure 4a, Table S2).

3.2.1 | Behavioural changes

One of the most commonly observed behaviours by fish to elevated suspended sediment is the avoidance of turbid water (Collin & Hart, 2015), an effect that has been observed in juvenile Coho salmon (*Oncorhynchus kisutch*, Salmonidae), Arctic grayling (*Thymallus arcticus*, Salmonidae), and Rainbow trout (*Oncorhynchus mykiss*, Salmonidae) (Newcombe & Jensen, 1996), species that have adapted to a range of environments. Avoidance behaviour (response type 1) can be induced at very low levels of suspended sediment (Figure 4a), but ceases once the disturbance is removed, or if the fish becomes acclimated (Berg, 1983; Berg & Northcote, 1985). Increased turbidity has also produced long-term shifts in local abundance and community composition. For example, a switch in dominance occurred between Common dab (*Limanda limanda*, Pleuronectidae) and European plaice (*Pleuronectes platessa*, Pleuronectidae) when turbidity increased as dredging escalated in the Dutch Wadden Sea over several years (De Jonge, Essink,

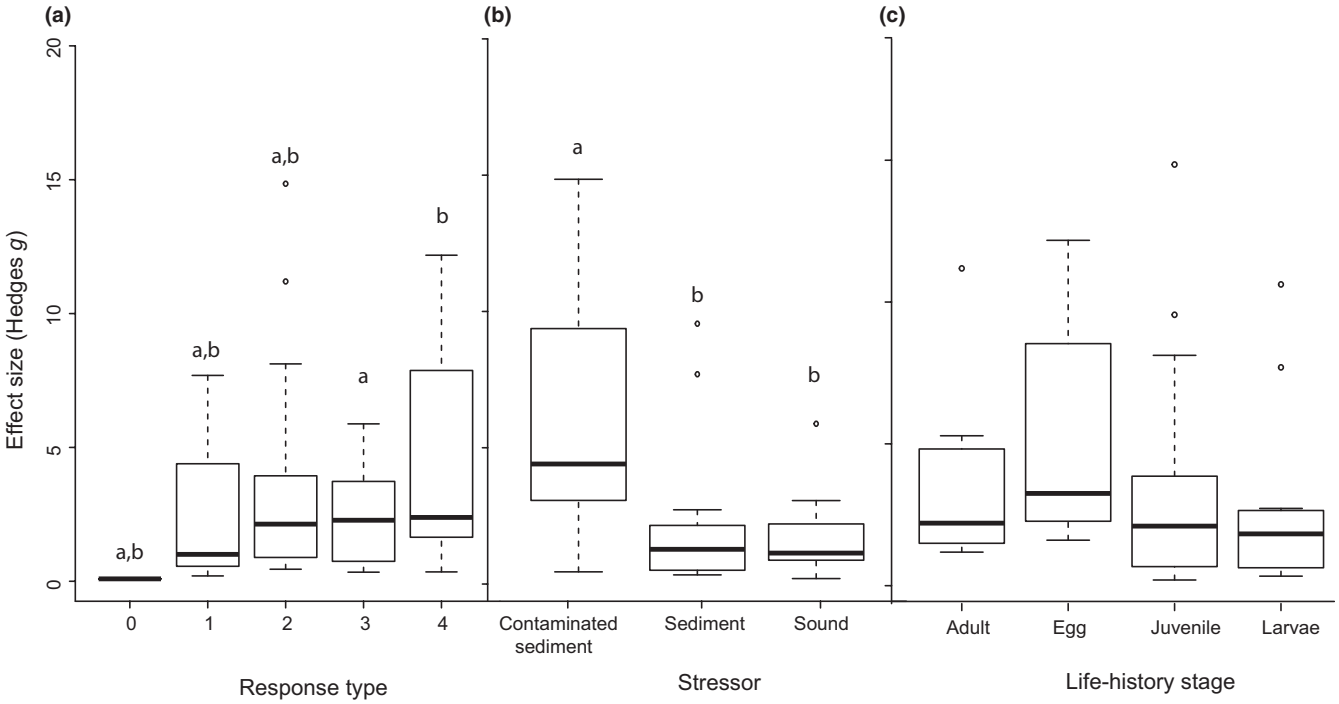

FIGURE 2 The impact of (a) response type, (b) stressor type, (c) life-history stage on effect size across all stressors. A response type of 0 = no effect, 1 = minor behavioural changes, 2 = minor physical damage or moderate behavioural changes, 3 = physiological impacts and 4 = increased mortality. Variables with non-overlapping letters above them are significantly different

& Boddeke, 1993). Additionally, the disappearance of mackerel in the Sea of Marmara, a key spawning ground for this species, was attributed to the presence of dredged material (Appleby & Scarratt, 1989); however it is likely that substantial changes in community composition are a direct result of long or frequent exposure.

Avoidance of dredged areas from dredging-related habitat modifications (e.g. sediment accumulation or loss) by fish can have a negative impact on fisheries at a local scale. For example, large deposits of dredged material in the Gulf of Saint Lawrence, Canada, were linked to a 3–7-fold decrease in catch per unit effort (CPUE) of Atlantic sturgeon (*Acipenser oxyrinchus*, Acipenseridae) (Hatin, Lachance, & Fournier, 2007). A reduced CPUE was related to either or both avoidance and a decreased effectiveness of fishing gear for species that visually locate bait (Utne-Palm, 2002). Conversely, CPUE can increase in turbid water if fish had a decreased ability to avoid fishing gear (Speas *et al.*, 2004). The return of fish to an area after a disturbance is highly dependent on the recovery of the environment to pre-disturbance conditions, the availability of alternative suitable habitat and the ecological plasticity of that species. Trade-offs between the risks associated with the disturbed environment and habitat and food availability will dictate the significance of behavioural changes brought on by dredging (Pirotta *et al.*, 2013).

Because turbidity often impairs visual acuity, activities and processes that require vision can be inhibited, leading to behavioural responses other than avoidance. Coral-associated damselfish were unable to locate live coral in turbid water, a process that relies on both visual acuity and chemoreception (O'Connor *et al.*, 2015; Wenger, Johansen, & Jones, 2011). This is particularly important for species with a pelagic larval phase, whereby the ability to find suitable habitat is crucial for development and survival during the very early life-history stages. If individuals settle into suboptimal habitat, they are more vulnerable to predation and experience slower growth rates (Coker, Pratchett, & Munday, 2009; Feary, McCormick, & Jones, 2009) which may have significant flow-on effects for the adult population (Wilson *et al.*, 2016). Once a fish has settled, however, their home range often expands to include a broader array of habitat patches and exploitable resources, thereby offsetting poor habitat choice at settlement (Wilson *et al.*, 2008). However, for one ubiquitous coral reef fish, the Lemon damselfish (*Pomacentrus moluccensis*, Pomacentridae), usually found in "clear lagoons and seaward reefs" (Syms & Jones, 2000), elevated suspended sediment reduced post-settlement movement by half (Wenger & McCormick, 2013). Fish that are unable to utilize the full extent of their home range due to elevated suspended sediment experience fitness consequences through a reduction in foraging and territorial defence (Lewis, 1997; Lönnstedt & McCormick, 2011). The meta-analysis indicated that many species exhibited moderate behavioural responses at concentrations as low as 20 mg/L, regardless of their habitat of origin, suggesting that dredging is likely to produce significant behavioural modifications.

3.2.2 | Effects on foraging and predation

It is already well established that foraging in both planktivorous and piscivorous fish is negatively affected by suspended sediment and that sedimentation affects herbivory (Utne-Palm, 2002). Foraging by planktivorous and drift feeding species is inhibited by reducing the reactive distance and the visual acuity of individual fish (Asaeda, Park,

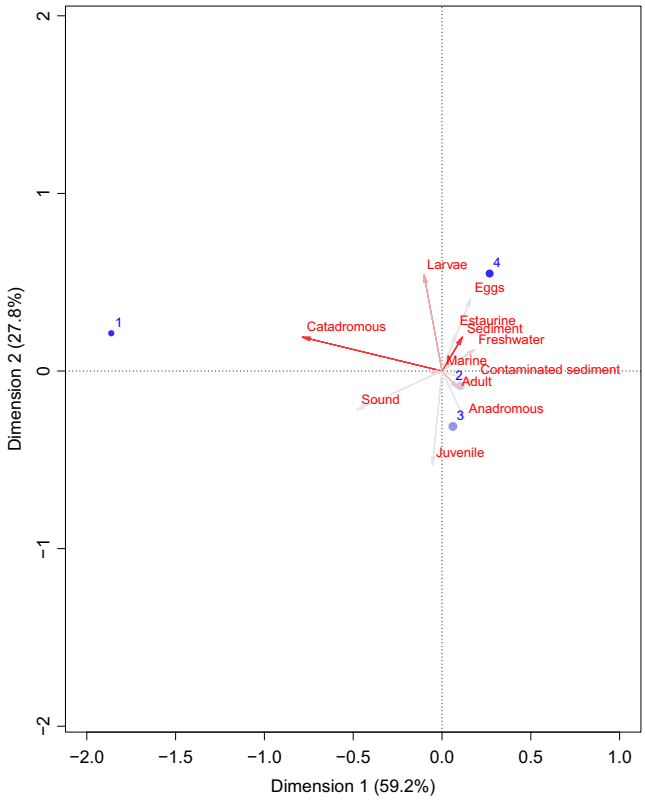

FIGURE 3 An asymmetric graph of the linear correspondence analysis, with the response type in the principal coordinates and the explanatory variables in reconstructions of the standardized residuals (square root of the relative frequency). Response type is represented by points, and the explanatory variables are represented by arrows. Point and vector shading intensity corresponds to the absolute contributions of the data to the display. Point size represents the relative frequency of each response type. The results indicate that across all stressors, larvae and eggs were most closely associated with lethal impacts (noted as 4), while catadromous fishes were most closely associated with behavioural effects (noted as 1). [Colour figure can be viewed at wileyonlinelibrary.com]

& Manatunge, 2002; Barrett, Grossman, & Rosenfeld, 1992; Gardner, 1981; Sweka & Hartman, 2003; Zamor & Grossman, 2007). Foraging success typically declines at higher levels of turbidity (Johansen & Jones, 2013; Utne-Palm, 2002). Berg (1983) documented a 60% reduction in prey consumed by Coho salmon in highly turbid water. Mild levels of turbidity, however, can sometimes enhance the contrast of plankton against its background, making it easier for planktivores to detect their prey (e.g. Utne-Palm, 1999; Wenger *et al.*, 2014). Some freshwater species such as the Rosyside dace (*Clinostomus funduloides*, Cyprinidae), Yellowfin shiner (*Notropis lutipinnis*, Cyprinidae) and Brook trout (*Salvelinus fontinalis*, Salmonidae) have shown an ability to cope with changing levels of turbidity by shifting their foraging strategies under conditions of high turbidity (30-40 NTU; Hazelton & Grossman, 2009; Sweka & Hartman, 2001). The Tenpounder (*Elops machnata*, Elopidae), for example, switches from fast-moving prey, such as fish, to slow-moving zooplankton when in a turbid estuary setting (Hect & Van der Lingen, 1992).

Although the literature has focused on the effects of suspended sediment on foraging, sedimentation can also inhibit foraging ability in benthic feeding species. For example, sediment embedded in algal turfs suppresses herbivory on coral reefs, with sediment removal resulting in a twofold increase in feeding by many herbivorous fish species (Bellwood & Fulton, 2008). Feeding intensity may also be influenced by sediment characteristics, with some parrotfish (*Scarus rivulatus*) displaying lower feeding rates when sediments were coarse and organic content was low (Gordon, Goatley, & Bellwood, 2016). Importantly, reduced feeding due to experimentally elevated sediment loads has been observed across different reef habitats, regardless of the natural sedimentation levels (Goatley & Bellwood, 2012). Ultimately, any reduction in foraging success leads to changes in growth, condition and reproductive output. Sweka and Hartman (2001) showed growth rates of Brook trout (*S. fontinalis*, Salmonidae) declined as turbidity increased (up to 40 NTU), due to an increase in energy used to forage. Similarly, increasing levels of suspended sediment reduced growth and body condition of the Spiny chromis (*Acanthochromis polyacanthus*, Pomacentridae) such that mortality increased by 50% in the highest suspended sediment concentrations (180 mg/L, Wenger, Johansen, & Jones, 2012).

Piscivores are especially sensitive to increasing turbidity because many are visual hunters that detect prey from a distance. An increase in suspended sediment reduces both light and contrast, decreasing encounter distances between predator and prey (Fiksen, Aksnes, Flyum, & Giske, 2002). Accordingly, several studies have shown a linear or exponential decline in piscivore foraging success with increasing turbidity (e.g. De Robertis, Ryer, Veloza, & Brodeur, 2003; Hect & Van der Lingen, 1992; Reid, Fox, & Whillans, 1999). The influence of turbidity on predation is, however, inconsistent among species. Turbidity had no effect on the predation rates of juvenile salmonids by Cutthroat trout (*Oncorhynchus clarkia*, Salmonidae; Gregory and Levings 1996), and Wenger, McCormick, McLeod, and Jones (2013) found a nonlinear

TABLE 4 A summary of the statistical outputs, including Rosenthal's fail-safe number, mean effect size, Wilks's lambda and the results of the linear correspondence analysis

Stressor	Rosenthal's fail-safe number	Mean effect size (Hedges' *g* ± *SE*)	Wilks's lambda (linear discriminant analyses)	Pr(>Chisq) (linear correspondence analysis)
All stressors	NA	NA	NA	.01
Suspended sediment	2,870	1.53 ± 0.33	<.0001	.303
Contaminated sediment (PAHs only)	246	4.24 ± 0.50	.41	.06
Sound	88	1.7 ± 0.5	.67	.23

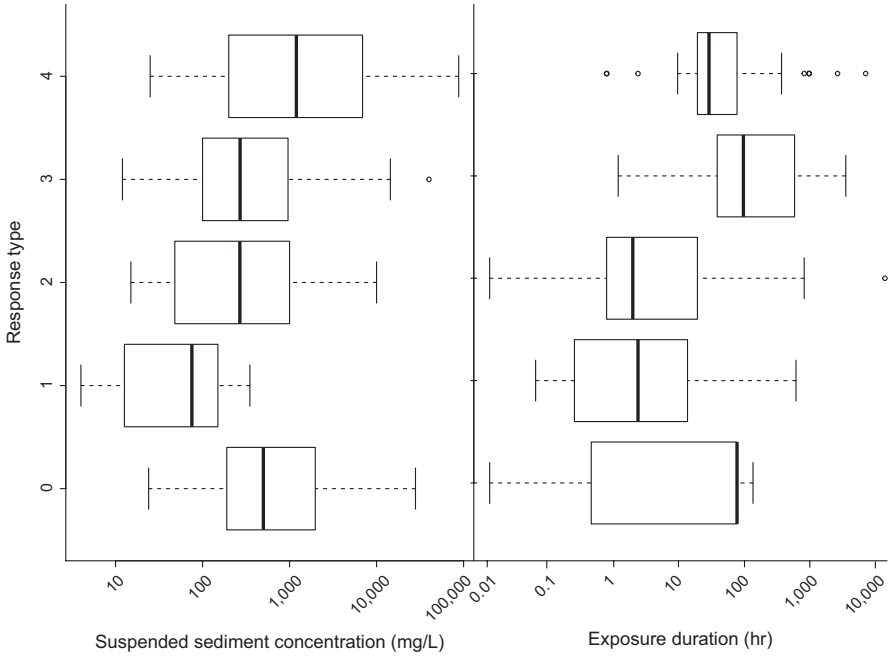

FIGURE 4 The impact of (a) suspended sediment concentration and (b) exposure duration on the type of effect elicited by suspended sediment. A response type of 0 = no effect, 1 = minor behavioural changes, 2 = minor physical damage or moderate behavioural changes, 3 = physiological impacts and 4 = increased mortality

relationship between increasing turbidity and predation success of dottybacks (*Pseudochromis fuscus*, Pseudochromidae), with intermediate levels of turbidity enhancing predation rates and high levels of turbidity reducing predation rates. The variation in species sensitivity to suspended sediment is reflected in the range of suspended sediment concentrations that elicited a reduced foraging and sublethal responses (Figure 4a). These results indicate predation success is partially dependent on factors other than vision and is likely to vary among species depending on the prey type, their natural ambient environment and the senses used to locate prey. However, the meta-analysis found that neither sediment type nor habitat of origin significantly influenced the effect size or response type elicited by suspended sediment exposure, suggesting that there are other factors of influence that have not yet been revealed.

3.2.3 | Light attenuation

Sediment in the water column not only reduces visual acuity due to its physical presence, it can also cause substantial light attenuation that impacts visual acuity (Jones, Fisher, Stark, & Ridd, 2015; Vogel & Beauchamp, 1999). Lower light levels can reduce the reactive distance of fish independent of the presence of sediment in the water column. A drastic change in the reactive distance of Bluegill (*Lepomis macrochirus*, Centrarchidae) from ~26 to 3.5 cm when light was reduced from 10.8 to 0.70 lux (Vinyard & O'Brien, 1976). While the assumption might be that the effects of increased turbidity in combination with low light intensity would be additive, studies that have examined the effects of both light reduction and increased turbidity have found mixed results. Utne (1997) observed a reduced reaction distance for the Two-spotted goby (*Gobiusculus flavescens*, Gobiidae) in both reduced light levels (<5 μmol m^{-2} s^{-1}) and increased turbidity, but there was no additive effect when light and turbidity levels were covaried. In contrast, Vogel and Beauchamp (1999) observed an additive effect of turbidity and light on reactive distance in Lake trout (*Salvelinus namaycush*, Salmonidae). De Robertis *et al.* (2003) found that turbidity decreased prey consumption by juvenile Chum salmon (*Oncorhynchus keta*, Salmonidae) and Walleye pollock (*Theragra chalcogramma*, Gadidae) in high light intensity, but not at low light intensity. Conversely, Miner and Stein (1993) observed that when light intensity was high (>460 lux), food consumption of Bluegill (*L. macrochirus*) larvae increased as turbidity increased, whereas food consumption decreased as turbidity increased in low light conditions (<100–300 lux). Still other studies have found no relationship, positive or negative, between light intensity, turbidity and foraging ability (Granqvist & Mattila, 2004).

3.2.4 | Physiological changes

Suspended sediment from dredging operations can lead to wide-ranging physiological effects in exposed fish. Increasing exposure to suspended sediment causes damage to gill tissue and structure, including epithelium lifting, hyperplasia and increased oxygen diffusion distance in the Orange-spotted grouper (*Epinephelus coioides*, Serranidae) and the Orange clownfish (*Amphiprion percula*, Pomacentridae) (Au, Pollino, Shin, Lau, & Tang, 2004; Hess, Wenger, Ainsworth, & Rummer, 2015). Under these conditions, increased pathogenic bacteria were also observed in Orange clownfish, while Lowe, Morrison, and Taylor (2015) found an increased parasite load on the gills of the Pink snapper (*Chrysophrys auratus*, Sparidae). Any reduction in gill efficiency impairs respiratory ability, nitrogenous excretion and ion exchange (Appleby & Scarratt, 1989; Au *et al.*, 2004; Wong, Pak, & Liu, 2013). The size of the gills is proportional to the size of the fish, meaning that the spaces between lamellae are smaller in larvae. It is therefore likely that sediment can more easily clog the gills and reduce their efficiency in smaller fish and larvae (Appleby & Scarratt, 1989). Larger and more angular sediment particles are also more likely to lodge between

the lamellae and cause physical damage to gill tissues and function (Bash, Berman, & Bolton, 2001; Servizi & Martens, 1987); however, this trend was not clear in the meta-analysis, with sediment type not influencing effect size or response type. As larvae have much higher oxygen requirements than other life-history stages, any reduced efficiency in oxygen uptake could increase mortality or sublethal effects (Nilsson, Östlund-Nilsson, Penfold, & Grutter, 2007). This may explain why larvae were highly associated with lethal impacts (Figure 3).

Structural changes in gills elevate haematocrit, plasma cortisol and glucose levels, all of which are consistent with oxygen deprivation (Awata, Tsuruta, Yada, & Iguchi, 2011; Collin & Hart, 2015; Wilber & Clarke, 2001). Increased sedimentation and suspended sediment can also reduce the amount of dissolved oxygen in water, exacerbating the direct physical damage to gills (Henley, Patterson, Neves, & Lemly, 2000). The sublethal effects described here strongly influence growth, development and swimming ability, all of which may inhibit an individual's ability to move away from dredging operations and compound any physiological effects (Collin & Hart, 2015).

3.3 | The effects of released contaminants on fish

The influence of contaminated sediments has a greater impact on fish than either suspended sediments or sounds originating from dredging (Figure 2b). There is substantial evidence that direct exposure to contaminants negatively effects fish (Jezierska, Ługowska, & Witeska, 2009; Nicolas, 1999), so it is not surprising that contaminated sediment has a greater effect on fish than clean sediment (Figure 2b). Studies on the effects of contaminated sediment examined a range of life-history stages (n = 8, 18, 3 and 7 for adults, juveniles, larvae and eggs). Fish species in the studies included five anadromous species, three estuarine species, 16 freshwater species and 12 marine species. The most commonly reported contaminants reported were metals (n = 13), polycyclic aromatic hydrocarbons (PAHs; n = 9) and polychlorinated biphenyls (PCBs; n = 4). There were also multiple studies that examined sediment contaminated from multiple sources (n = 10; Table S3). The effects elicited from contaminated sediment were varied, with two studies showing no effect, one study observing behavioural changes, 11 studies recording physical damage, 15 studies recording physiological and sublethal impacts and seven studies documenting mortality. However, more than half of the studies on contaminated sediment effects on fish used sediment contaminated with multiple contaminants (n = 19/36), making quantitative comparison among studies problematic (Table S3). However, many of the studies collected sediment from polluted aquatic environments, indicating that dredging in polluted environments is likely to expose fish to multiple contaminants. There was only one study on heavy metals (cadmium), two studies on PCBs and six studies on PAHs where an effect size could be calculated that had test contaminants individually and that had units that could be compared. Effect sizes for studies on PAHs ranged from 2.83 to 6.20, with a mean effect size of 4.24 ± 0.50 (SE) (Table S3).

We conducted analysis only on the PAH studies given the low sample sizes of the other contaminant studies. None of the predictor variables (concentration, exposure duration, life-history stage, habitat and response type) in the linear mixed-effects model significantly influenced variation in effect size (Table 3). Rosenthal's fail-safe number for PAH studies was 246, whereas it was 14 for PCB studies (Table 4). Although this number is very low for PCB experiments, it is probably indicative of inadequate studies on the topic, rather than publication bias. Furthermore, the results of the linear correspondence analysis and the calculated chi-square statistic indicated that there was no significant association between the predictor variables (habitat and life-history stage) and response type elicited by exposure to sediment contaminated with PAHs (p = .06; Table 4).

The results of the linear discriminant analysis and the Wilks's lambda results indicated that PAH concentration and exposure times did not explain the response type elicited (p = .41; Table 4).

3.3.1 | Hydrophobic organic contaminants

The studies reviewed and synthesized suggest substantial impacts from exposure to sediment contaminated with hydrophobic organic chemicals (Table S3). Hydrophobic contaminants, such as legacy persistent organic pollutants (POPs; including PCBs, polybrominated diphenyl ethers [PBDEs], organochlorine pesticides OCPs, dioxins PCDDs, furans PCDFs) and high-molecular weight polyaromatic and aliphatic hydrocarbons (PAHs), are closely associated with organic material in sediments (Simpson *et al.*, 2005). Some form naturally and may be present in sites with no human impacts (some PAHs, dioxins and aliphatics; Gaus *et al.*, 2002). Others are only common in sediments exposed to shipping activity and/or industrial development (e.g. PCBs, organotins; Haynes & Johnson, 2000). Anthropogenic compounds with a high bioaccumulation potential (some PCB congeners, PCDDs, PBDEs) may be present in low to moderate concentrations in sediments even at sites well-removed from the source through water and aerial transport and deposition (Evers, Klamer, Laane, & Govers, 1993) or incorporated in the food web (Losada *et al.*, 2009; Ueno *et al.*, 2006). The release of hydrophobic organics requires desorption from particulates which can readily occur under certain environmental conditions (Bridges *et al.*, 2008; Eggleton & Thomas, 2004). The meta-analysis provides further support to the idea that desorption of hydrophobic organics can occur by showing that exposure to contaminated sediment results in a greater effect size than other dredging-related stressors. Further, Steuer (2000) found that around 35% of PCBs downstream of a riverine remedial dredging programme were in the dissolved fraction (i.e. had been released). Thus, exposure to these compounds should therefore not be ignored during the risk assessment process, even at capital dredging sites.

Johnson *et al.* (2014) comprehensively reviewed the direct impacts of POPs on fish and demonstrated the breadth of reproductive impacts on adults (e.g. steroidogenesis, vitellogenesis, gamete production or spawning success) as well as lethal and non-lethal developmental (spinal and organ development, growth) impacts on embryos and larvae. There is also potential for maternal transfer of POPs through accumulation in oocyte lipid stores and the impact of PAHs on steroidogenesis (Monteiro, Reis-Henriques, & Coimbra, 2000) and vitellogenesis (reviewed by Nicolas, 1999). Specific to crude oils, Carls *et al.* (2008)

demonstrated that toxicity to fish embryos was due to the dissolved PAH fraction. This implies that release of sediment-associated PAHs may cause similar deformities as those observed following exposure to oil. Any activity that exposes fish, regardless of its life stage, to POPs or PAHs should be considered high risk to animal health and, in exploited long-lived predators, a potential risk to human consumers. A full understanding of the sediment contaminant profile and release dynamics is required to fully protect fish stocks, particularly where ripening of spawning fish, or their eggs, embryos or larvae is likely to encounter POPs released through the resuspension of contaminated sediment, given the high sensitivity of larvae and eggs to dredging-related stressors (Figure 3).

3.3.2 | Metals

Metals in sediments are generally present as sulphides, a form generally not bioavailable and therefore non-toxic (Rainbow, 2007). Sediments rich in iron sulphides, however, have a large capacity to bind potentially toxic metals (e.g. copper, zinc, nickel, lead, cadmium) by exchanging the bound iron with the competitor metal (Rainbow, 1995). When iron sulphides are resuspended, they are readily oxidized, causing localized acidification, and release of bioavailable and toxic ionic metal (Petersen, Willer, & Willamowski, 1997). Some metals are released more readily than others (Maddock, Carvalho, Santelli, & Machado, 2007), so the duration for which the contaminated sediment is exposed to the seawater is a critical variable. Fine sediments (silts and clays) remain in suspension longer and will therefore release more metals.

It is clear that there is a gap in the understanding of the potential for metals adsorbed to sediment to be taken up by fishes. Despite the well-understood desorption of metals from sediment (reviewed by Eggleton & Thomas, 2004), only 12 studies have examined the effects of metal-contaminated suspended sediment on fish, with five of them focusing on single metals and only one where the effect size was able to be calculated. However, the limited laboratory studies that have investigated uptake have demonstrated that it can and does occur (Table S3). Further, the studies that examined sediment contaminated with multiple heavy metals highlight that exposure to metal-contaminated sediment can elicit large effects, regardless of the response type (Table S3).

Although not widely studied, it is possible to infer the likely impacts of the uptake of metals from contaminated suspended sediment based on a large body of empirical studies examining direct effects of metal exposure on fish. Metals impact reproductive output and early development in fish via a range of entry routes and mechanisms (reviewed by Jezierska *et al.*, 2009). Metals accumulate in gonad tissue (Alquezar, Markich, & Booth, 2006; Chi,, Zhu, & Langdon, 2007) and in the egg shell and chorion causing developmental delays, changes in time to hatch and larval deformities (Chow and Chang 2003; Witeska, Jezierska, & Chaber, 1995). Heavy metals such as mercury, zinc and cadmium are also known to reduce sperm motility (Abascal, Cosson, & Fauvel, 2007; Kime *et al.*, 1996). At higher but still within concentrations recorded in the environment (0.1 and 10 mg/L), ionic metals can be lethal to larvae (*Cyprinodon variegatus*, Cyprinidae; Hutchinson, Williams, & Eales, 1994). Jezierska *et al.* (2009) reviewed the physiological stress responses in adult fish exposed to ionic metals as osmoregulatory disturbance (copper), antioxidant inhibition (cadmium), interference with the citric acid cycle (cadmium), oxidative stress, disruption of thyroid hormones (lead) and antagonistic binding to oestrogen receptors (cadmium). With the wide range of known impacts of exposure to metals, full characterization of metals in sediment and release kinetics is required on a case-by-case basis to assess any exposure and impacts to fish.

3.4 | The effects of hydraulic entrainment on fish

Hydraulic entrainment, through the direct uptake of aquatic organisms by the suction field generated at the draghead or cutterhead during dredging operations (Reine *et al.*, 1998), results in the localized by-catch of fish eggs, larvae and even mobile juveniles and adults. A review of entrainment rates of fishes, fish eggs and fish larvae has been previously undertaken by Reine *et al.* (1998). However, as studies only record rates of entrainment, without controls for comparison, it was not possible to calculate effect sizes or conduct quantitative analyses. The studies did, however, record a variation in the mortality or damage that occurred and suggest that eggs are more vulnerable to entrainment than adults, with observed damage/mortality of 62.8 ± 13.6 (mean ± *SE*) for eggs compared to 38.4 ± 13.2 for adults (Table S4). This result, in combination with the results from the meta-analysis that demonstrate eggs and larvae are most likely to experience lethal impacts (Figure 3), underscores the vulnerability of early life-history stages to dredging.

3.4.1 | Entrainment of eggs and larvae

Most published research into the effects of dredging entrainment on fish eggs and larvae has been carried out in riverine or estuarine river systems (Griffith & Andrews, 1981; Harvey, 1986; Harvey & Lisle, 1998; Wyss, Aylin, Burks, Renner, & Harmon, 1999). Whereas extensive attention has been placed on the consequences of entrainment by hydropower facilities or power plant cooling water intakes, less research has been devoted to entrainment by hydraulic dredges. Because volumes of water entrained by dredges are small in comparison with these other sources, the entrainment rates of eggs and larval fish are generally thought to represent a minor proportion of the total fish production (Reine & Clarke, 1998; Reine *et al.*, 1998). Hydraulic dredging is not directly comparable to hydropower or cooling water sources in other ways. For example, trailer suction hopper dredges are mobile, generally advancing at speeds under several metres per second. Depending on the capabilities of a given dredge, pumping capacities span a very wide range. When entrainment occurs in close proximity to large spawning aggregations, however, replenishment of fish populations could theoretically be suppressed via the removal of reproductive adults. Where sufficient ecological information exists, the risk of entraining larval fish and eggs can be minimized by restricting dredging during key reproductive and recruitment time periods (Suedel, Kim, Clarke, & Linkov, 2008) and avoiding nurseries

and spawning aggregations. While the entrainment rates are likely to represent a small proportion of total larval production, fish entrained at the egg, embryo and larval stages will experience extremely high mortality rates (Harvey & Lisle, 1998; Table S4), although mortality rates will vary among fish species and development stages (Griffith & Andrews, 1981; Wyss *et al.*, 1999).

3.4.2 | Entrainment of mobile juvenile and adult fish

Documented entrainment rates of mobile fish species are low, but are highest for benthic species or those in high densities (Drabble, 2012; Reine *et al.*, 1998). While the potential for entrainment of abundant demersal species can be relatively high, the overall mortality rates of entrained fish may be low. Mortality rates vary depending on the type and scale of dredging operation, with the longer term survival of fish after entrainment reliant on the method of separation of the dredged sediment from the fluid, and on how the dredged sediment is disposed (Armstrong, Stevens, & Hoeman, 1982). For example, mortality rate of estuarine fish in Washington immediately after hydraulic entrainment and deposition into the hopper was 38%, but was 60% for pipeline dredges with a cutter head (Armstrong *et al.*, 1982). In the English Channel, only six of the 23 adult fish entrained by a suction trailer dredger were damaged (Lees, Kenny, & Pearson, 1992; Table S4). Furthermore, as fish may avoid areas that are repeatedly dredged (Appleby & Scarratt, 1989), hydraulic entrainment may be more pronounced during capital dredging, when fish densities have not yet been altered by coastal development.

3.5 | Effects of dredging sounds on fish

Sound levels recorded from dredge operations ranged from 111 to 170 dB re 1 μPa rms, with exposure lasting from 2 min to 10 days (Table S5). There were seven records each on the effects of sound on both juvenile and adult fish, one record for larvae and one unknown. There were two studies on catadromous fish, one on an estuarine fish, eleven records from freshwater species and two from the marine environment (Table S5).

There was a range of endpoints measured and responses elicited from dredge sound, although none of these were lethal. Five studies observed behavioural changes (response type 1), six studies recorded physical damage and substantial behavioural changes (response type 2), and five studies measured physiological stress (response type 3). Effect sizes ranged from 0.2 to 5.9, with a mean effect size of 1.7 ± 0.5 (*SE*) (Figure 2b; Table 2).

According to the results of the generalized linear mixed-effects model, only response type had any significant influence on the effect size from dredge sound ($p = .03$; Table 3), with effect size generally increasing as the severity in response increased (Table S5). However, there was no lethal response recorded in any of the studies we reviewed. The other predictor variables tested were decibel level, exposure duration, life-history stage and habitat. Rosenthal's fail-safe number was 88, indicating that our results are not an artefact of publication bias (Table 4).

The results of the linear correspondence analysis and the calculated chi-square statistic indicated that there was no association between the predictor variables (habitat, life-history stage and species) and response type elicited by exposure to continuous sound ($p = .23$). Similarly, according to the linear discriminant analysis, neither decibel level or exposure duration drove variations in response type ($p = .67$; Table 4).

While the effects of anthropogenic sound on fish have been thoroughly reviewed by Hawkins, Pembroke, and Popper (2015) and Popper and Hastings (2009) and synthesized into guidelines by Popper *et al.* (2014), they do not specifically include dredging as a sound source. Moreover, there is a paucity of information on the impacts of anthropogenic sound on fish in terms of their physiology and hearing. Data exist for only ~100 of the more than 32,000 recorded fish species (Popper & Hastings, 2009). Based on the existing information, underwater noise can affect fish in a number of ways, including (i) behavioural responses, (ii) masking, (iii) stress and physiological responses, (iv) hearing loss and damage to auditory tissues, (v) structural and cellular damage of non-auditory tissues and total mortality, (vi) impairment of lateral line functions and (vii) particle motion-based effects on eggs and larvae (Popper & Hastings, 2009; Popper *et al.*, 2014; Table S4).

Effects of dredging noise vary among fish species with one of the most important determinants being the presence or absence of a swim bladder (Popper *et al.*, 2014), which we did not account for in the meta-analysis. Fish species that have a swim bladder used for hearing are more likely affected by continuous noise than those without a swim bladder (Popper *et al.*, 2014). For example, after exposure to white noise at 170 dB re 1 μPa rms for 48 hr, goldfish (*C. auratus*, Cyprinidae) developed temporary loss of sensory hair bundles and experienced a temporary threshold shift (TTS, i.e. temporary hearing loss) of 13–20 dB (Smith, Coffin, Miller, & Popper, 2006; Table S5), enough to change their ability to interpret the auditory scene. After 7 days, TTS had recovered, and after 8 days, hair bundle density had recovered (Smith *et al.*, 2006). In another study, exposure to 158 dB re 1 μPa rms for 12 and 24 hr resulted in TTS of 26 dB in goldfish and 32 dB in catfish (*Pimelodus pictus*, Pimelodidae) (Amoser & Ladich, 2003; Table S5). Hearing thresholds recovered within 3 days for the goldfish, and after 14 days for catfish, and the duration of exposure had no influence on long-term hearing loss (Amoser & Ladich, 2003). The results of the meta-analysis support this observation, with exposure duration having no impact on the response type elicited by sound.

Several published studies exist that have quantified dredging sounds from hydraulic and mechanical dredging (e.g. Reine, Clarke, & Dickerson, 2014; Reine, Clarke, Dickerson, & Wikel, 2014; Thomsen, McCully, Wood, White, & Page, 2009). The available evidence indicates that dredging scenarios do not produce intense sounds comparable to pile driving and other in-water construction activities, but rather lower levels of continuous sound at frequencies generally below 1 kHz. However, when dredging includes the removal or breaking of rocks, the sound generated is likely to exceed the sound of soft sediment dredging. The exposure to dredging sounds does depend on site-specific factors, including bathymetry and density stratification of the water column (Reine, Clarke, & Dickerson, 2014). Exposures to a

given sound in relatively deep coastal oceanic waters will be different to those experienced in shallow estuaries with complex bathymetries. While sound levels produced by dredging can approach, or exceed, the levels tested in the aforementioned studies, received sound levels will be lower than source levels (Reine, Clarke, & Dickerson, 2014). As sound pressure is significantly lower from natural sources compared to that produced by anthropogenic impacts such as dredging, most fish species do not have the physiology to detect sound pressure (Hawkins *et al.*, 2015; Popper *et al.*, 2014) and therefore show no TTS in response to long-term noise exposure (Popper *et al.*, 2014). Impacts on fish from dredging-generated noise are therefore likely to be TTSs (temporary hearing loss) in some species, behavioural effects and increased stress-related cortisol levels (Table S4). Finally, although dredging may not cause levels of sound that can be physiologically damaging to fish, dredging noise may mask natural sounds used by larvae to locate suitable habitat (Simpson *et al.*, 2005).

4 | SUMMARY AND RECOMMENDATIONS

Increased waterborne trade and the expansion of port facilities infer that dredging operations will continue to intensify over the next few decades (PIANC 2009). The development of meaningful management guidelines to mitigate the effects of dredging on fish requires a thorough understanding of how dredging can impact fish. This review represents a substantive descriptive and quantitative assessment of the literature to characterize the direct effects of dredging-related stressors on different life-history stages of fish. Across all dredging-related stressors, studies that reported fish mortality had significantly higher effect sizes than those that describe physiological responses, although indicators of dredge impacts should endeavour to detect effects before excessive mortality occurs. Our results demonstrate that contaminated sediment led to greater effect sizes than either clean sediment or sound, suggesting additive or synergistic impacts from dredging-related stressors. Importantly, we have explicitly demonstrated that early life stages such as eggs and larvae are most likely to suffer lethal impacts, which can be used to improve the management of dredging projects and ultimately minimize the impacts to fish. Although information on drivers of effect sizes provides insight into the factors contributing to impacts, an examination of the drivers that influence the elicited response type is more informative to management, because it allows for early detection of stress, which can trigger management intervention before sublethal and lethal impacts occur. As such, this review provides critical information necessary for dredging management plans to minimize impacts from dredging operations on fish. Furthermore, it highlights the need for *in situ* studies on the effects of dredging on fish which consider the interactive effects of multiple dredge stressors and their impact on sensitive species of ecological and fisheries value.

Currently, the literature on dredging-related stressors is biased towards examining the effects of suspended sediment, as is evidenced by the large number of studies that exist on the topic compared to other stressors. While suspended sediment is a ubiquitous stressor in any dredging project, our review highlights the need for further research on how contaminants released during dredging, noise associated with dredging and hydraulic entrainment can impact fish. There is also a paucity of direct field measurements of the effects of dredging on fish, which needs to be addressed. The characterization of multiple, long-term impacts from stressors associated with dredging needs to consider all combinations of acute toxicity, chronic stress, loss of habitat and the frequency and duration of repeated exposures. This is particularly important in the light of the results that contaminated sediment caused significantly higher effect sizes than sediment alone, which suggests there are additive or synergistic impacts occurring. An increased understanding of how each stressor acts alone or in combination will improve our ability to effectively manage potential impacts from dredging.

In many developed countries, the disposal of contaminated sediments is well regulated and includes strict requirements to avoid contamination of the environment, as the release of contaminants into the water column can cause environmental damage (Batley and Simpson 2009). The release of contaminants from sediments resuspended during dredging and their impact on fish depend on the characteristics of the sediment, water chemistry, suspension time and the compound itself (reviewed by Eggleton & Thomas, 2004). Because seldom is only one contaminant found in contaminated sediment, systematic studies on the effects of combined contaminants should be carried out to better assess the potential impact to fish of dredging-induced exposure to contaminated sediments. Where the contaminant load is significant and results in the slow leaching of toxins, the re-establishment of habitat and appropriate larval settlement sites could be significantly prolonged. Repeat maintenance dredging of contaminated sediments will expose resident fish populations to multiple pulses of SS and released toxicants. While the impact of a single exposure may have little or no effect, repeated exposures or the effects of exposure of fishes to multiple contaminants can cause contaminant accumulation to levels that are toxic (Maceda-Veiga *et al.* 2010).

Although the effects of suspended sediment, noise, hydraulic entrainment and contaminant release have been considered separately here, there are likely to be interactions among dredging-related stressors that could reduce or magnify the intensity of a response or raise or lower the threshold of response. Interactive effects of multiple stressors on fish are poorly represented in the literature. Crain, Kroeker, and Halpern (2008) performed an analysis of 171 fully factorial studies using two stressors on marine organisms or communities finding that the overall impact of two stressors tends to be synergistic in heterotrophs, which the results of this meta-analysis support. However, the interactions may present themselves differently. For instance, where high-molecular weight hydrophobic contaminants and metals co-occur in sediments and resuspension, the combination of the particular compounds needs to be considered in determining risk, because of potential toxicity across all life-history stages. In this case, reducing the concentration or exposure to contaminated sediment is likely to be the best management option. Conversely, the identification of larvae and eggs as being more vulnerable to dredging-related stressors, as demonstrated by the meta-analysis, suggests that dredging management aimed at minimizing dredging activities during certain times of

year when eggs and larvae would be abundant would be warranted. Given the complexities of different dredging-related stressors and their influence on the response type and size of effect elicited, it is likely that more than one management intervention would be necessary. This review provides critical information about factors influencing how fish would respond to dredging.

This review has assessed the weight of evidence that exists for direct effects of dredging on fish. However, indirect effects on fish through loss of prey, changes to biochemical processes and habitat loss may also occur. In particular, changes to habitat may be substantial and could exceed the impacts caused by direct effects of dredging-related stressors on fish (Barbier *et al.*, 2011). Consequently, benthic habitats have been explicitly accounted for in management recommendations and plans (Erftemeijer *et al.*, 2013; PIANC 2009). When fish are considered in dredging management plans, there is often limited scientific evidence used to support the recommended management interventions (Dickerson, Reine, & Clarke, 1998; Suedel *et al.*, 2008). The information generated in this meta-analysis demonstrates that there can also be significant direct effects of dredging on fish, which can compound the indirect effects of habitat loss, leading to further impacts. Therefore, management plans should consider both indirect and direct impacts to fish, in line with the precautionary principle.

The knowledge generated here represents a rigorous assessment of the available information, especially in relation to suspended sediment. However, it highlights the current lack of *in situ* data that are critical to the decision-making process for environmental impact assessments. There is a great need for more applied research to provide the necessary information to management agencies so that they can make educated decisions on the impacts of future dredging developments to fish and fishery resources in freshwater, estuarine and coastal ecosystems. In particular, targeted Before, After, Control, Impact ("beyond" BACI) designed *in situ* field studies focused on assessing multiple responses of key and representative species (across all life-history stages) to multiple stressors over time are needed. Such studies would be challenging both financially and logistically, but if conducted in collaboration with dredging companies, they could provide a realistic experiment of dredging impacts and ultimately reduce costs of dredging operations and environmental impacts. We recommend that managers use the information generated here in tandem with any information on the effects of dredging on critical fish habitat, in order to develop comprehensive practices to target direct and indirect impacts.

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AUTHOR CONTRIBUTIONS

All authors presented at or contributed to a workshop on effects of sediment on fish held at the University of Western Australia, led by EH. AW conducted all of the analyses, and AW, EH, CR, SW, SN, DC, BS, NB, PE and DM wrote the review. SW, JM, JH, MD and RE edited the final document.

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SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

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