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Effects of dredging-related activities on finfish: a review and management strategies

Euan Harvey[1,8], Amelia Wenger[2], Benjamin Saunders[1,8], Stephen Newman[3,8], Shaun Wilson[4,8], Mike Travers[3,8], Nicola Browne[1], Chris Rawson[1,8], Douglas Clarke[5], Jean Paul Hobbs[1,8], Jennifer McIlwain[1,8], Richard Evans[4], Paul Erftemeijer[7,8], Dianne Mclean[6,8], Martial Depczynski[7,8]

[1] *Department of Environment and Agriculture, Curtin University, Bentley, Western Australia, Australia*

[2] *School of Geography, Planning, and Environmental Management, University of Queensland, St. Lucia, Queensland, Australia*

[3] *Department of Primary Industries and Regional Development, North Beach, Western Australia, Australia*

[4] *Department of Biodiversity, Conservation and Attractions, Perth, Western Australia, Australia*

[5] *HDR, Vicksburg, MS, United States*

[6] *School of Plant Biology and Oceans Institute, The University of Western Australia, Crawley, Western Australia, Australia*

[7] *Australian Institute of Marine Science, Perth, Western Australia, Australia*

[8] *Western Australian Marine Science Institution, Perth, Western Australia, Australia*

WAMSI Dredging Science Node

Theme 8 Report

Projects 8.1 & 8.2

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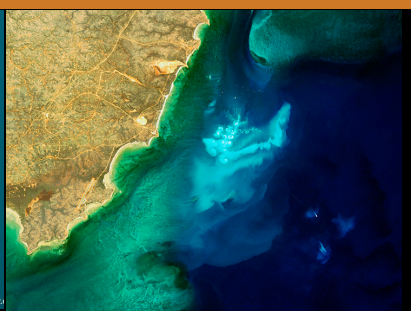

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WAMSI Dredging Science Node

The WAMSI Dredging Science Node is a strategic research initiative that evolved in response to uncertainties in the environmental impact assessment and management of large-scale dredging operations and coastal infrastructure developments. Its goal is to enhance capacity within government and the private sector to predict and manage the environmental impacts of dredging in Western Australia, delivered through a combination of reviews, field studies, laboratory experimentation, relationship testing and development of standardised protocols and guidance for impact prediction, monitoring and management.

Funding Sources

The $20million Dredging Science Node is delivering one of the largest single issue environmental research programs in Australia. This applied research is funded by **Woodside Energy, Chevron Australia, BHP Billiton and the WAMSI Partners** and designed to provide a significant and meaningful improvement in the certainty around the effects, and management, of dredging operations in Western Australia. Although focussed on port and coastal development in Western Australia, the outputs will also be broadly applicable across Australia and globally.

This remarkable **collaboration between industry, government and research** extends beyond the classical funder-provider model. End-users of science in regulator and conservation agencies, and consultant and industry groups are actively involved in the governance of the node, to ensure ongoing focus on applicable science and converting the outputs into fit-for-purpose and usable products. The governance structure includes clear delineation between end-user focussed scoping and the arms-length research activity to ensure it is independent, unbiased and defensible.

And critically, the trusted across-sector collaboration developed through the WAMSI model has allowed the sharing of hundreds of millions of dollars worth of environmental monitoring data, much of it collected by environmental consultants on behalf of industry. By providing access to this usually **confidential data,** the **Industry Partners** are substantially enhancing WAMSI researchers' ability to determine the real-world impacts of dredging projects, and how they can best be managed. Rio Tinto's voluntary data contribution is particularly noteworthy, as it is not one of the funding contributors to the Node.

Funding and critical data **Critical data**

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Corresponding author and Institution: Euan S. Harvey (Curtin University). Email: euan.harvey@curtin.edu.au

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Front cover images (L-R)

Image 1: Trailing Suction Hopper Dredge *Gateway* in operation during the Fremantle Port Inner Harbour and Channel Deepening Project. (Source: OEPA)

Image 2: Impacts to species like the Pink Snapper (*Pagrus auratus*), that aggregate to spawn at known times and locations can be avoided by scheduling dredge programs to avoid peak spawning periods. (Photo: Reef Life Survey)

Image 3: Dredge Plume at Barrow Island. Image produced with data from the Japan Aerospace Exploration Agency (JAXA) Advanced Land Observing Satellite (ALOS) taken on 29th August 2010.

Image 4: Of the 13 species assessed the Western Blue Groper (*Achoerodus gouldii*) was identified as the highest priority for management and research into the effects of dredging. (Photo: Euan Harvey)

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Executive Summary

Dredging can have significant impacts on marine environments, but its effects on fishes are still largely unresolved. One strategy that has been used to minimise the impacts of dredging on corals is to cease, or limit dredging activity during sensitive life history periods (Critical Environmental Windows[1]) which are most vulnerable to the mechanisms of dredging. A workshop in 2013 was held to develop the structure for a literature review which assessed the effects of dredging on different life stages of marine fishes from around the world, and to determine if there were any clear Environmental Windows for fish. This review identified a suite of traits which make fishes vulnerable to dredging. Increases in suspended sediments can affect processes requiring vision such as foraging, hunting and predator avoidance. Increases in suspended sediment can also have physiological impacts such as gill impairment causing decreases in respiration and increases in diseases. It can also impact chemoreception and the ability of larval recruits to detect suitable habitats for settlement. Benthic eggs and larvae can be smothered, producing negative impacts on survival and growth of recruits, while pelagic eggs can be impacted by sediment adhesion resulting in the eggs sinking. Hydraulic entrainment of fishes at various life stages and their food sources is a localised concern, as is the marine noise associated with dredging. As a consequence of these potential impacts on fishes the impacts of dredging are of concern to recreational and commercial fisheries, traditional owners and environmental managers.

Data were collated from peer reviewed literature, grey literature and expert opinion on the traits of a broad range of temperate and tropical Western Australian marine fishes, focussing on those that are targeted by commercial and recreational fishers. The traits that were assessed were:

- economic significance;
- recreational significance;
- customary significance;
- social significance;
- IUCN/EPBC significance;
- reproductive strategy;
- recruitment;
- distribution/endemism;
- connectivity; and
- habitat specificity.

From this information we have developed a priority assessment process that can be used to identify individual species which may be vulnerable to dredging activities, or are higher priority for management and research. This process has teased out what is known and assisted in identifying gaps in the current knowledge of fishes in Western Australia. Large gaps in local knowledge exist, particularly with respect to the timing of key ecological processes for the majority of species of marine fishes and their preferential spawning and recruitment habitats. Of the 102 species we considered, there was some information on spawning and recruitment habitat for 41 and 16 of those species respectively. Research should be prioritised to address important temporal and spatial processes, i.e. identifying spawning and recruitment periods, and the identification of the critical habitats that

[1] EPA (2016) Technical Guidance: Environmental Impact Assessment of Marine Dredging Proposals. Environmental Protection Authority, Western Australia

support these processes. We recommend further targeted research for high priority species of fish that may be vulnerable to dredging in order to assess the ecological impacts and contribute to informed management.

The current knowledge of the spawning periods of Western Australian fishes indicates that the peak spawning period for both temperate and tropical species occurs in the warmer months between October and April. However, fish spawn throughout the year, and in any given month at least 20% of fish species may be spawning, whether in temperate or tropical Western Australia. To limit the effect of dredging impacts on spawning to 50% of fish species or fewer, dredging could be avoided during the peak spawning window between October and April. The degree of seasonality appears to be more pronounced in tropical waters with approximately 80% of tropical species spawning during summer (December-February) compared to approximately 65% in temperate waters during the same period. The proportion of tropical species spawning in winter (June-August) reduced to approximately 20% whereas approximately 30-45% of temperate species were recorded as spawning during this period. However, the patterns illustrated are limited by a lack of fundamental knowledge for many species, and it is likely that, with further research, the peak spawning periods, especially for tropical species, will be extend over longer periods of time.

There are also opportunities to spatially manage dredging and disposal to minimise impact during the critical early life history stages of fish where critical habitats that support spawning aggregations or the processes of spawning and recruitment are known. Critical habitat information has been summarised from the limited information available. Inshore, and nearshore habitats are critical for spawning and recruitment in many fish species, and these areas may coincide with the location of dredging projects. Consideration of both temporal and spatial critical windows may be necessary on a case by case basis.

We have collated and analysed the available literature on the effect of sediment concentrations on fishes from a variety of aquatic habitats. The impacts of very fine sediment on fishes range from minimal impacts at 10 mgL^{-1} to extreme at 1,000 mgL^{-1}. This information can be used to derive critical threshold concentrations for very fine suspended sediment that, if not exceeded, would afford a degree of protection for a proportion of fish species. For example, the analysis suggests that a trigger value set at 166 mgL^{-1} would protect 25% of fish species, while a trigger value of 2.4 mgL^{-1} would protect 99% of fish species. Approximately 50% of fish species would be protected if very fine sediment concentrations were maintained below 80 mgL^{-1}. Based on the available information, there is 95% confidence that 90% of fish species would be protected if suspended sediment concentrations did not exceed 10 mgL^{-1}. A precautionary approach should be used with the implementation of any management based upon these trigger values, due to the variety of habitat types and limited data on which the analysis was based. Only 4 of the 17 species used in the analysis (24%) are present in Western Australia, and 10 of the 16 genera (63%). In addition, the studies on fish used in the analysis had a range of exposure times and differing endpoints, varying from behavioural changes to lethal effects. The values used in our analysis are the concentrations where no effect of sediment was detected for the measured endpoint. Therefore, it is possible that other effects of sediment may have occurred, but were not measured in these studies.

Once the vulnerability of the local fish species, and the consequences of any impact to those species has been assessed, strategies can be implemented to manage this risk. Such management strategies may involve restrictions on dredging if endangered or high consequence species are present. However, management strategies to address lower consequence species should also be considered. This could involve the removal, relocation, and or restoration of vulnerable species and habitats. Site attached species, or those with small home ranges are more vulnerable to the effects of dredging or habitat loss. Many of these fish species, and the habitats that they rely on, are of value to the aquarium trade and it may be possible to collect them from the vicinity of proposed dredging works. This may be facilitated through preferential harvesting prior to impact, and may relieve collection pressure on stocks elsewhere. Alternatively, or in addition, many habitat associated species with small home ranges may be relocated, along with their habitat. Successful relocation also opens up the possibility of restoration of the affected area in the future, by returning part of the original community.

Considerations for predicting and managing the impacts of dredging

The review and case study highlight four important directions for management of the effects of dredging on fishes; temporal management, spatial management, management of sediment concentration, and management of vulnerable species. Details of these management implications are outlined below. The review also found that the critical impacts of dredging will be on fish eggs and larvae, suggesting management should focus on actions that protect early life history processes such as spawning and recruitment. However, there is a lack of knowledge on many of the effects of dredging on fishes, including a fundamental lack of ecological and biological information for many Western Australian fish species. It is possible to consolidate the available information into implications and recommendations for management, though there is also a need for targeted research to fill critical knowledge gaps for vulnerable species of high conservation, ecological, and/or fisheries value.

EPA (2016) [1] provides a framework for the management and prediction of the environmental impacts of dredging in Western Australia. This framework has three phases into which the new information provided through this review could be integrated:

- a pre-development phase, which includes surveys and investigations to define the system in which dredging might occur;
- an impact assessment phase, in which the potential dredging-generated pressure fields and their effect on sensitive components of the environment need to be predicted, and monitoring undertaken; and
- a post-assessment verification of the impacts and gathering of reference data.

In the following sections we discuss the findings of this review in the context of management implications during each of these phases.

Pre-development Surveys

Pre-development surveys are recommended to identify the species and abundances of fish present, and the types of habitat that are available within the proposed area. On the basis of this information, *a-priori* assessment of risk could be made to inform dredge scheduling and management. This section focusses on how the information presented within this review can be used in conjunction with the results of pre-development surveys to inform spatial and temporal management of the impacts of dredging on fishes. Fish species lists compiled during pre-development surveys, or from existing historical data sets, could be used in conjunction with the information compiled in appendix 4, to identify species specific critical windows during which dredging could be avoided or managed. These windows may be temporal, linked to the timing of critical processes, or spatial, encompassing critical habitats or bioregions – or a combination of the two.

An *a-priori* assessment of critical habitats should be undertaken for each new area to be developed. Dredging and disposal should be spatially managed to minimise impact during the critical early life history stages of fish. Where possible, critical habitats that support spawning aggregations or the processes of spawning and recruitment should be protected from the effects of dredging and disposal. Recent research has highlighted the significance of nearshore habitats as nursery areas. As such, there is a risk to population recruitment for some species as most dredging programs occur near shore, increasing the potential to impact critical nearshore habitats for recruitment or spawning. Western Australian researchers are continuing to work towards identifying the nearshore habitats that are important for different life history stages of fish species[2]. As this research is completed it will identify critical recruitment habitat types that could be avoided if possible, either spatially or temporally.

It is important to note that an effect of dredging during the key ecological processes of spawning and/or recruitment may have consequences that are not detectable in the fish assemblages for a number of years after the disturbance has occurred. In addition to a general decline in abundance, an impact during spawning or

[2] Wilson, S.K. Personal communication. Western Australian Department of Parks and Wildlife, Perth, Western Australia

recruitment may become apparent over time as a missing cohort in one or two species. For this reason, it is suggested that pre and post-development monitoring take an assemblage wide approach, and collect information on the size-class frequency of the species recorded at sites that are predicted to be affected and unaffected by dredging.

Impact Prediction

We have collated and analysed the available literature on the effect of sediment concentrations on fishes from a variety of aquatic habitats. Fish species that populate naturally turbid marine environments are very poorly represented in the available literature (but see appendix 2, Box 2 for a recent example). Despite being based upon species from a range of aquatic habitats, the limited available data are in general agreement and show some consistent patterns. The impacts of very fine sediment on fishes range from minimal impacts at 10 mgL^{-1} to extreme at 1,000 mgL^{-1} (appendix 1, Figure 3). This information can be used to derive critical threshold concentrations for very fine suspended sediment concentrations. A trigger value set at 166 mgL^{-1} would protect 25% of fish species, while a trigger value of 2.4 mgL^{-1} would protect 99% of fish species (appendix 1, Figure 3). Approximately 50% of fish species would be protected if very fine sediment concentrations were maintained below 80 mgL^{-1}. Based on the available information, there is 95% confidence that 90% of fish species would be protected if suspended sediment concentrations did not exceed 10 mgL-1 (appendix 1, Figure 3). If any of these trigger values were to be implemented a precautionary approach should be used. This is because the studies on which our analysis was based use species from a variety of habitat types, and only 4 of the 17 species (24%) and 10 of the 16 genera (63%) used in the analysis are present in Western Australia. In addition, the studies had a range of exposure times and differing endpoints, varying from behavioural changes to lethal effects. The values used in our analysis are the concentrations where no effect of sediment was detected for the measured endpoint. Therefore, it is possible that other effects of sediment may have occurred, but were not measured in these studies. If the use of trigger values is adopted ongoing monitoring would be required, with an immediate management response to reduce sediment concentrations once the threshold value is exceeded.

The current knowledge of the spawning periods of Western Australian fishes indicates spawning for both temperate and tropical species occurs predominantly between October and April with a more pronounced late spring/summer peak evident for tropical species. However, the patterns illustrated are limited by a lack of fundamental knowledge for many species, and it is likely that, with further research, the peak spawning periods, especially for tropical species, will be extend over longer periods of time. Current research indicates that many tropical species spawn repeatedly throughout the year, or over extended periods of 9–10 months[3]. Rather than increasing susceptibility to dredging, species that maintain such extended breeding seasons may be resilient to short term impacts that affect only a small proportion of their breeding cycle. However, in any given month of the year greater than 20% of the fish species present may be spawning, whether in temperate or tropical Western Australia (appendix 1, Figure 2). In acknowledgement of the uncertainty, and to limit the effect of dredging on spawning to 50% of fish species or fewer, temporal management strategies should focus on the peak spawning window between October and April (appendix 1, Figure 2).

Management

The potential effects of dredging on fish species is highly variable, as critical life history processes are highly variable in space and time. For example, pink snapper (*Chrysophrys auratus*) spawn between April and October in the sub-tropical Gascoyne coast bioregion (latitude 26 °S). In the west coast bioregion around the Perth metropolitan area (32 °S), spawning occurs from August to January[4]. Cockburn Sound, Warnbro Sound, and Owen Anchorage are key spawning locations for this species, and pink snapper within Warnbro and Cockburn

[3] Wakefield, C.B. et al. Unpublished data. Western Australian Department of Fisheries, Perth, Western Australia

[4] Wakefield CB, Potter IC, Hall NG, Lenanton RCJ, Hesp SA (2015) Marked variations in reproductive characteristics of snapper (*Chrysophrys auratus*, Sparidae) and their relationship with temperature over a wide latitudinal range. ICES Journal of Marine Science: Journal du Conseil 72:2341-2349

sounds are currently protected from fishing between October and January to protect spawning aggregations. Further south in the Albany region (35°S), spawning takes place over 2–3 months between September and December[4]. The Shark Bay world heritage area in the Gascoyne coast bioregion is a unique location for this species. Shark Bay supports three different pink snapper stocks with high genetic diversity, and each is managed separately. Pink snapper in the eastern gulf spawn between May and July, and are currently protected from fishing during this time[5]. Within Denham Sound spawning also occurs between May and July. However, in the Freycinet estuary area spawning occurs between August and October[5], and the area is closed to fishing from August 15th to September 30th. Water temperature is a key trigger for spawning in this species, with spawning occurring at water temperatures of between 19°C and 21°C[4]. To minimise the impacts of dredging on pink snapper spawning, dredging programs could be scheduled to avoid the peak spawning periods, which differ throughout the distribution of this species.

The variation in the timing of critical life history processes within and between species highlights the need for location specific research into the reproductive biology of key fish species. Such information is needed to inform decisions on when dredging is most appropriate. In an effort to prioritise management or conservation actions we have developed an approach using a vulnerability × consequence score to assess the susceptibility of species to the impacts of dredging (appendix 2, Table 1). This could be used to assess the vulnerability of fish species on a case by case basis. Management of the vulnerability of species in a particular area, or for a particular project should begin with an *a-priori* assessment of the species present, and critical habitats for recruitment or spawning. The information compiled in appendix 4 provides a valuable tool that summarises the current state of knowledge on the spawning and recruitment timing, duration and habitats for many fish species in Western Australia. This tool can be used to identify species-specific temporal or spatial critical windows where dredging could be avoided. An example of a species for which the timing and locations of the critical processes of spawning and recruitment are known is the Australian herring (*Arripis georgianus*). This species spawns between April and June with a peak in late May through to early June. It is a broadcast spawner that spawns only in Western Australia. It forms multiple small spawning aggregations throughout the west coast bioregion. To negate the effects of dredging on Australian herring spawning, dredging programs in the west coast bioregion could be scheduled to avoid the months of April until June. Alternatively, to minimize any effects during peak spawning the period from mid-May until mid-June could be avoided. Australian herring is one of the few Western Australian species for which information on pelagic larval duration and time until settlement is known. Larvae are transported south by the Leeuwin current, and recruitment occurs into shallow, vegetated habitats along the south coast of Western Australia up to 4 months after spawning, between June and September. Critical windows to wholly protect the spawning and recruitment processes for this species might encompass the temporal window from April until September, and nearshore habitats within the west and south coast bioregions. In a species such as Australian herring which has a relatively narrow spawning period, the vulnerability to pressures during that period are high. But the spatial extent of the spawning and the threatening process (in this case dredging) would also be considered and reconciled during any risk assessments undertaken. If spawning is latitudinally broad and the pressure is latitudinally narrow then the cumulative risk at the population level is reduced. So an individual dredging event may place low pressure on that year's spawning. However if there are multiple dredging activities that are spatially separate but temporally aligned then the cumulative pressure on the fish stock may be significant.

Species within the same family, and coexisting in similar areas may require different management strategies. For example, the western blue groper (*Achoerodus gouldii*) and the baldchin groper (*Choerodon rubescens*) are two high scoring Labrid species which were identified as high concern through the vulnerability x consequence rating (appendix 2, Table 1). Both of these species spawn in inshore habitats in the west and south coast bioregions, with the baldchin groper having a more northern distribution and extending into the Gascoyne coast bioregion. However, they have quite different temporal critical windows (appendix 4). The western blue groper spawns

[5] Jackson G, Norriss JV, Mackie MC, Hall NG (2010) Spatial variation in life history characteristics of snapper (*Pagrus auratus*) within Shark Bay, Western Australia. New Zealand Journal of Marine and Freshwater Research 44:1-15

over a period of 5 months between June and October, while the baldchin groper spawns over a seven month period from July through January, with a peak spawning period between September and January. Further examples within other families can be found within appendix 4. For example, the King George whiting (*Sillaginodes punctata*) and the southern school whiting (*Sillago bassensis*) also show contrasting patterns. Both species inhabit the west coast and south coast bioregions, and spawn in and recruit into nearshore and inshore habitats. However, they have opposing temporal critical windows. The King George whiting has a peak spawning period in winter (June and July), while the southern school whiting spawns throughout the year with a peak in summer (December through March). Recruitment for these species follows a similar pattern, with peak recruitment for the King George whiting into estuarine and nearshore habitats between September and April, followed by a peak recruitment period between November and April into nearshore habitats for the southern school whiting. Given these examples, and the information presented in appendix 4, the management of the effects of dredging on critical processes for different species would likely require very different strategies. A tool such as the vulnerability x consequence rating (appendix 2, Table 1) may be useful to prioritise species for targeted management strategies.

Once the vulnerability of the fish species present, and the consequences of any impact to those species has been assessed, strategies can be implemented to manage the risk. Such management strategies may involve restrictions on dredging if endangered or very high consequence species are present. However, management strategies to address lower consequence species should also be considered. This could involve the removal, relocation, and or restoration of vulnerable species and habitats. Site attached species, or those with small home ranges are more vulnerable to the effects of dredging or habitat loss (appendix 4). For example, the red clownfish (*Amphiprion rubrocinctus*) was assessed as having the highest vulnerability score due to its specific and limited habitat requirements (appendix 2, Table 1). Many similar fish species, and the habitats that they rely on, are of value to the aquarium trade and it may be possible to harvest them from the vicinity of proposed dredging works. This may be facilitated through preferential harvesting prior to impact, and may relieve collection pressure on stocks elsewhere. Alternatively, or in addition, many habitat associated species with small home ranges may be relocated, along with their habitat. Successful relocation also opens up the possibility of restoration of the affected area in the future, by returning part of the original community.

Post-assessment

In summarising the current knowledge of the effects of dredging on fishes, the state of knowledge of critical processes in Western Australian fishes, and developing an approach that may be used to prioritise management and future research this project has focused on the pre-development and impact prediction phases. However, post-development surveys of the fish assemblages upon completion of a dredging project provide an opportunity to assess the extent and severity of any actual impacts that may occur – particularly when there are good data on the spatial and temporal aspects of the suspended sediment fields that occurred during the dredging campaign. The results of surveys conducted at the completion of dredging at control and impact sites would provide valuable information on the effects of the particular pressure field (expressed as TSS intensity, duration and frequency) on the numbers and distributions of juvenile and adult fishes in the area of interest. These surveys should take an assemblage wide approach collecting information on the abundance and size-class frequency of the species recorded for a number of years after dredging has finished. With comparison to the data collected during pre-development surveys this may allow for the measurement of any effect during the key ecological processes of spawning and/or recruitment.

Residual knowledge gaps

There have been some studies on the effects of suspended sediment on fish. However, there remain key knowledge gaps on the effects of different concentrations and exposure durations on different life history stages of fish. These gaps must be addressed to ensure that trigger values that initiate management and timing of critical environmental windows are robust. Finally, a management priority should be to address the fundamental

knowledge gaps that exist for a large proportion of Western Australian fish species. The vulnerability x consequence score (appendix 2, Table 1) can be extended using the information presented in appendix 4 to prioritise species for future research efforts. Research should be prioritised to address important temporal and spatial processes, i.e. identifying spawning and recruitment periods, and the identification of critical habitats that support these processes. A concerted effort towards the consolidation of grey literature and unpublished data collections may begin to fill these knowledge gaps.

1 Introduction and background

Dredging is a critical component of most major marine infrastructure developments in Western Australia and around the world. Dredging is the excavation and relocation of benthic material from lake, river or sea beds, and is commonly used to improve navigable depths for shipping channels and harbours. Dredging operations for the extraction of oil and liquid gas off the north-west coast of Western Australia have expanded since the mid-2000s (Masini et al. 2011). This is combined with an unprecedented expansion of port facilities in the region (Hanley 2011). A description of dredging within a Western Australian context can be found in the Technical guidance on Environmental Impact Assessment of Marine Dredging Proposals (EPA 2016).

Despite the necessity of dredging, several studies have documented multiple potential stressors associated with dredging activities. Chief among these are sediment stress (suspended and deposited), release of toxic contaminants, hydraulic entrainment and noise pollution (appendix 1, Figure 1; Reine & Clarke 1998; Wilber & Clark 2001; Reine et al. 2014a, b; McCook et al. 2015). There is a growing body of literature that examines how dredging-associated stressors impact fish (e.g. Utne-Palm 2002; Suedel et al. 2008; Wenger et al. 2014), yet few, if any, global standards exist that protect finfish of conservation, ecological or fishery importance from dredging practices. Even in jurisdictions where environmental protection has been a high priority, with well-developed robust regulatory controls (e.g. Australia, United States and European Union), debate continues on the application of management practices intended to mitigate dredging-induced impacts (Dickerson et al. 1998; NAS 2001; Suedel et al. 2008). This is due to the uncertainties that exist in assessing the scales of dredging impacts, particularly with respect to the multitude of tolerance thresholds displayed by marine fish. Despite the significant dredging operations undertaken across a range of marine environments, our knowledge of the relationships between dredging-related pressures and fish is poor. Such knowledge is essential for predicting potential impacts and designing appropriate management strategies (NAS 2001; PIANC 2010; Kemp 2011). 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, or best management practices (NAS 2001; PIANC 2009). Despite known impacts from dredging (Erftemeijer & Lewis 2006; Erftemeijer et al. 2012), the extent of environmental damage associated with dredging operations is often unclear due to a poor understanding of cause-effect relationships between environmental conditions and organism responses. In addition, the long-term consequences are poorly understood as most impact studies occur during the period of dredging, and two to three years post-dredging. However, impacts that result from altered benthic conditions may persist for much longer. The limited knowledge is a direct result of large spatial and temporal differences in intrinsic conditions, sediment plume characteristics and species responses to sediment loads.

The effects described above can be divided into two broad categories, direct effects and indirect effects (appendix 1, Figure 1). Direct effects are those that are caused as a result of the dredging process, and on fishes may include:

1) Noise (as a result of dredging machinery and shipping)
2) Entrainment (within the dredge footprint,)

Indirect effects are those that occur as a result of the disturbance of the seabed and suspension of sediment within the water column. These may include:

3) Indirect effects on fishes of loss of habitat/prey (within the dredge footprint and disposal ground footprint)
4) Elevated suspended sediment concentrations
5) Increased light attenuation
6) Increased sediment deposition
7) Release of legacy contaminants or toxicants

1.1 Dredging in Western Australia

Several large-scale marine infrastructure developments have, or will commence (subject to relevant approvals) in the lower Kimberley, Pilbara, mid-west, south-west and south coast regions of the state (Masini et al. 2011). The volume of dredged materials associated with these projects is large on a global scale at approximately 200 million cubic metres of sediment, with individual projects capable of influencing areas in excess of 1000 square kilometres (Masini et al. 2011). Despite the significant extent of dredging operations in a range of environments around the world, knowledge about relationships between dredging-related pressures and fish is poor. In Western Australia observations of fish assemblages before and after dredging is only available for one dredging project (see Box 1: Western Australian Case Study). Such knowledge is essential for predicting impacts and designing appropriate management strategies. In Western Australia, hydraulic dredging (hopper and cutterhead) is predominantly used for large dredging projects (EPA 2016). This technique mixes large volumes of water with sediments to form a slurry that is either pumped through a pipeline to an upland location, into a hopper bin or is sidecast away from the dredging site (Morton 1977). The rate of the cutterhead rotation, the vertical thickness of the dredge cut and the swing rate of the dredge all affect the amount of suspended sediments and turbidity levels created by the dredging project (LaSalle 1990).

1.1 Legislative framework influencing dredging activities

Within Western Australia the Technical guidance on Environmental Impact Assessment of Marine Dredging Proposals (EPA 2016) provides a framework for the development of monitoring and management plans for dredging proposals. A similar framework is also applied elsewhere in Australia. Frameworks such as EPA (2016) aim to deliver a spatially consistent description of the extent, severity and duration of the predicted impacts and the timeframes over which they will recover. In Western Australia the Environment Protection Authority (EPA) conducts an environmental impact assessment of dredging programs that are likely to have a significant impact on the environment (as defined in the *Environmental Protection Act 1986*). However, not all dredging proposals in Western Australia are subject to assessment by the EPA, and some may be considered by other agencies or authorities (EPA 2016).

In Australia the over-arching pieces of legislation which govern dredging are the *Environment Protection (Sea Dumping) Act 1981*, the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) and the *Great Barrier Reef Marine Park Act 1975* (Department of Environment 2009). The Sea Dumping Act implements Australia's obligations under the 1996 London Protocol. The EPBC Act 1999 is the Australian Government's central piece of environmental legislation. It provides a legal framework for the environmental assessment and approvals process for actions that are likely to have a significant impact on matters of national environmental significance (www.environment.gov.au 2015a). In addition, dredging associated with offshore petroleum and greenhouse gas storage activities in Commonwealth waters require assessment and authorisation under the Offshore Petroleum and Greenhouse Gas Storage (OPGGS) Act and associated regulations, including the OPGGS (Environment) Regulations 2009 (www.nopsema.gov.au 2015). Applications under these regulations are submitted through the Australian commonwealth agency, the National Offshore Petroleum Safety and Environmental Management Authority (NOPSEMA). For proposed dredging activities that occur within three nautical miles from the Territorial sea baseline, State and Northern Territory Governments are primarily responsible for regulating loading and dumping activities (Department of Environment 2009). The onus is on the applicant to identify and comply with all relevant legislation including State and Territory Law, where applicable. In most instances, the States and the Northern Territory have requirements under their own legislation for dredging and dumping that occurs within their adjacent coastal waters to three nautical miles. There remains a requirement for approval under the EPBC Act when dredging and dumping activities that occur within waters within the limits of a State or the Northern Territory are likely to have an impact on a matter of national environmental significance (Department of Environment 2009). The Australian Government is in the process of instituting a 'One Stop Shop' for environmental approvals that will accredit environmental assessment and approval processes under national environment law, and remove the need for duplicate approval processes (www.environment.gov.au 2015b).

Box 1: Western Australian Case Study

Barrow Island Gorgon Project Dredging Program Marine Monitoring

The Gorgon Project at Barrow Island in the Pilbara region of north-western Australia was the largest single resource project in Australia's history, and one of the world's largest natural gas projects. The capital dredging program associated with the Liquefied Natural Gas Jetty and Materials Offloading Facility had an anticipated dredge volume of approximately 7.6 million m^3 (Chevron Australia Pty Ltd 2011).

Baseline and post-development surveys were undertaken to assess change in the structure of fish assemblages following the dredging works (Chevron Australia Pty Ltd 2013). These surveys used baited remote underwater stereo-video systems (stereo-BRUVs). Baseline surveys were conducted during October 2008 and March 2009, while post development surveys were conducted in November 2011, December 2012, and November 2013. These surveys were conducted across four different zones (Zones of High Impact, Zones of Moderate Impact, Zones of Influence and Reference) a design similar to that suggested in the environmental assessment guideline for marine dredging proposals (EPA 2016). An assessment of changes in the fish assemblage structure from baseline surveys to post-development surveys was conducted for each zone (Chevron Australia Pty Ltd 2013).

Despite the scale of the dredging operations, measurable changes in the fish assemblage were found only within the Zones of High Impact, and the Zones of Moderate Impact, and only for some species (Chevron Australia Pty Ltd 2013). A pulse decline in mean species richness was recorded at impact sites associated with the materials offloading facility and liquefied natural gas jetty during the first post development survey in 2011, however during the second post-development survey in 2012 the mean species richness had recovered to baseline levels (Chevron Australia Pty Ltd 2013). A similar pattern was observed for three of the indicator species for reef fish assemblage, bluespotted tuskfish (*Choerodon cauteroma*), blackspot tuskfish (*Choerodon schoenleinii*), and coral trout (*Plectropomus* spp). Between the baseline and first post-development survey in 2011 a decrease in the mean abundance of bluespotted tuskfish (40%) and blackspot tuskfish (38% decrease) was detected in coral habitat in the impact zones (Zones of High Impact, Zones of Moderate Impact, and Zones of Influence considered together), but both had recovered to baseline levels (30% and 35% increase respectively) by the second post-development survey in 2012 (Chevron Australia Pty Ltd 2013). The mean abundance of coral trout declined (92% decrease) in coral habitat in the Zones of High Impact from baseline to the first post-development survey. But, during the second post development survey the abundance of coral trout had increased slightly, so that it was no longer significantly lower than during the baseline surveys (Chevron Australia Pty Ltd 2013).

A decrease in the mean abundance of purple threadfin bream (*Pentapodus emeryii*), was found between baseline and both the 2011 (82% decrease from baseline) and 2012 (65% decrease from baseline) post-development surveys in coral habitat within the Zones of High Impact (Chevron Australia Pty Ltd 2013). There was no sign of a recovery of this species (Chevron Australia Pty Ltd 2013). The report describing the outcomes of the third post-development survey in 2013 has not yet been publicly released, so the latest recovery trends are unknown.

It should be noted that a constraint of the use of the zoning system recommended in the environmental assessment guide for marine dredging proposals (EPA 2016) to monitor fish species is the increasing area that is encompassed by each zone moving further from the impact. The zones of highest impact are by necessity smaller than the zones of influence and reference areas. When monitoring mobile fauna, such as fishes, the small size of the highest impact zones reduce the ability to gather multiple independent replicates, and thus potentially the power to detect change, or to generalise patterns throughout the zone. For example, in the case study described here only two sites are located within the Zone of High Impact associated with the materials offloading facility and liquefied natural gas jetty, one in coral habitat and one in sand/sessile invertebrates (Chevron Australia Pty Ltd 2013). Where feasible, these highest impact zones should be large enough to incorporate multiple sites in each habitat while maintaining spatial independence.

The majority of the changes described in this case study were detected within the Zone of High Impact closest to the materials offloading facility and liquefied natural gas jetty. That this is the area from which material was directly dredged suggests that the most influential factor on the fish assemblages may be habitat removal, rather than turbidity or suspended sediment. The recovery of species that was observed within these zones during the 2012 post-development survey, might conceivably be attributed to the presence of new complex structure within the zone as the materials offloading facility and liquefied natural gas jetty were constructed.

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1.2 International legislation

Internationally, most dredging activities fall under the umbrella of the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (the London Convention 1972) and the later Protocol to the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (the London Protocol 1996) (www.imo.org 2015). Under the 1996 protocol dumping of dredged material is allowed, providing permits are obtained. Underneath this agreement dredging is regulated by layers of legislation. A national or regional over-arching legislation is underpinned by state and local legislations. Dredging programs are dealt with on a case by case basis.

The United States of America are signatories of the London Protocol. Beneath this, there are several layers of relevant regulatory controls, both at the state and federal levels. For a dredging project to proceed the applicant must first receive a permit for any activity occurring in state waters, which extend out three miles from ocean shorelines. The regulatory offices of the Army Corps of Engineers issue the required permits, but must stipulate any compliance controls specified by the state (www.usace.army.mil/ 2015). For example, in a regulatory document known as a Water Quality Certificate (WQC) the state may impose an environmental window to protect an individual species of concern, or an assemblage of fishes, such as migratory species. Without a WQC a project cannot proceed. In all cases the states coordinate with the appropriate federal agencies, primarily the National Marine Fisheries Service (NMFS) and US Fish and Wildlife Service, but also the US Environmental Protection Agency if contaminated sediments are involved. A number of Federal legislative actions overlap within state waters, but cover US territorial waters as well. The NMFS is mandated to protect fishery resources under the Magnuson-Stevens Fishery Conservation and Management Act, originally passed in 1976 and amended and reauthorized periodically since (www.nmfs.noaa.gov/ 2015b). In 1996 Congress amended the Act to identify Essential Fish Habitat (EFH), which extends protection to “waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity” (www.nmfs.noaa.gov/ 2015c). When protection involves a species of Threatened or Endangered status, as defined by the Endangered Species Act (ESA) of 1973, the controls deemed appropriate by the NMFS and the US Fish and Wildlife Service take precedence over all other considerations (www.nmfs.noaa.gov/ 2015a). Any finding of potential significant impact will lock in all “reasonable and prudent” measures to ensure protection of the species (www.nmfs.noaa.gov/ 2015a). Management under the ESA focuses on protection of individuals, whereas EFH and other protections occur at the population level.

Within the European Union there are a number of pieces of legislation that govern any proposed dredging programs. In addition, many of the European states are signatories to the London Protocol. The European Council Water Framework Directive (Directive 2000/60/EC) is the framework legislation for the protection of surface and ground water in the EU. The aim of this directive is to achieve ‘Good Status’ of European waters (ec.europa.eu 2015b). The Marine directive (2008/56/EC) is closely linked with the Water Framework Directive, and aims to achieve ‘Good Environmental Status’ of European marine waters (ec.europa.eu 2015d). Dredging programs also come within the Waste Framework Directive (2008/98/EC) (ec.europa.eu 2015e), and where dredging may threaten wetlands or wild birds within the Habitat Directive (92/43/EEC) and the Birds Directive (2009/147/EC) (ec.europa.eu 2015a, c). The Birds Directive bans activities that directly threaten birds (ec.europa.eu 2015a). The European states are obligated to develop and implement their own legislation to ensure compliance with these directives. For example, in England and Wales licences must be obtained for all marine dredging programs (www.gov.uk 2015b). Licences are issued by the Marine Management Organisation and compliance with the relevant EU framework is assessed during the licencing process (www.gov.uk 2015a).

1.3 Objectives of the project and structure of report

1.3.1 Objectives

This report covers the findings of Projects 8.1 and 8.2. The objective of Project 8.1 was to review existing literature on the impacts of dredging on critical environmental windows of finfish in both tropical and temperate areas using the steps outlined below.

- identify critical ecological processes (such as spawning, recruitment and early post- recruitment growth) and associated environmental windows;
- identify pressure parameters and the intensity associated with dredging;
- assess likely sensitivities/tolerances of the ecological processes and windows to dredge related activities; and
- provide recommendations on key areas of concern and potential mitigation.

The objective of Project 8.2 was to assess the information presented in the literature review and to identify (by a workshop) gaps in the knowledge and highlight priority areas for future laboratory/field experiments.

1.3.2 Structure

This project is reported in two main sections:

Section 1 – A review of the effects of dredging on fishes (Part 2 of this report); and

Section 2 – a framework for the development of management strategies within a Western Australian Context (Part 3 of this report).

Part 2 reviews the existing literature on the impacts of dredging on finfish in both tropical and temperate areas. Part 3 grew out of the workshop which was held in October 2013. It integrates the current knowledge of the key ecological windows around spawning and recruitment for Western Australian Fish species, and provides a prioritisation tool which can be used to assess the vulnerability of species to dredging, or to direct further research efforts.

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ORIGINAL ARTICLE

WILEY FISH and FISHERIES

A critical analysis of the direct effects of dredging on fish

Amelia S. Wenger[1] | Euan Harvey[2,3] | Shaun Wilson[2,4,5] | Chris Rawson[2,3] | Stephen J. Newman[2,3,6] | Douglas Clarke[7] | Benjamin J. Saunders[2,3] | Nicola Browne[3] | Michael J. Travers[2,3,6] | Jennifer L. Mcilwain[2,3] | Paul L. A. Erftemeijer[4] | Jean-Paul A. Hobbs[2,3] | Dianne Mclean[4] | Martial Depczynski[2,4,8] | Richard D. Evans[4,5]

[1]School of Earth and Environmental Sciences, University of Queensland, St. Lucia, Qld, Australia

[2]The Western Australian Marine Science Institution, Floreat, WA, Australia

[3]Department of Environment and Agriculture, Curtin University, Bentley, WA, Australia

[4]School of Plant Biology and Oceans Institute, the University of Western Australia, Crawley, WA, Australia

[5]Marine Science Program, Science and Conservation Division, Department of Parks and Wildlife, Kensington, WA, Australia

[6]Western Australian Fisheries and Marine Research Laboratories, Department of Fisheries, Government of Western Australia, North Beach, WA, Australia

[7]HDR, Vicksburg, MS, USA

[8]Australian Institute of Marine Science, University of Western Australia, Crawley, WA, Australia

Correspondence
Amelia S. Wenger, School of Earth and Environmental Sciences, University of Queensland, St. Lucia, Qld, Australia.
Email: Amelia.wenger@gmail.com

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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

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

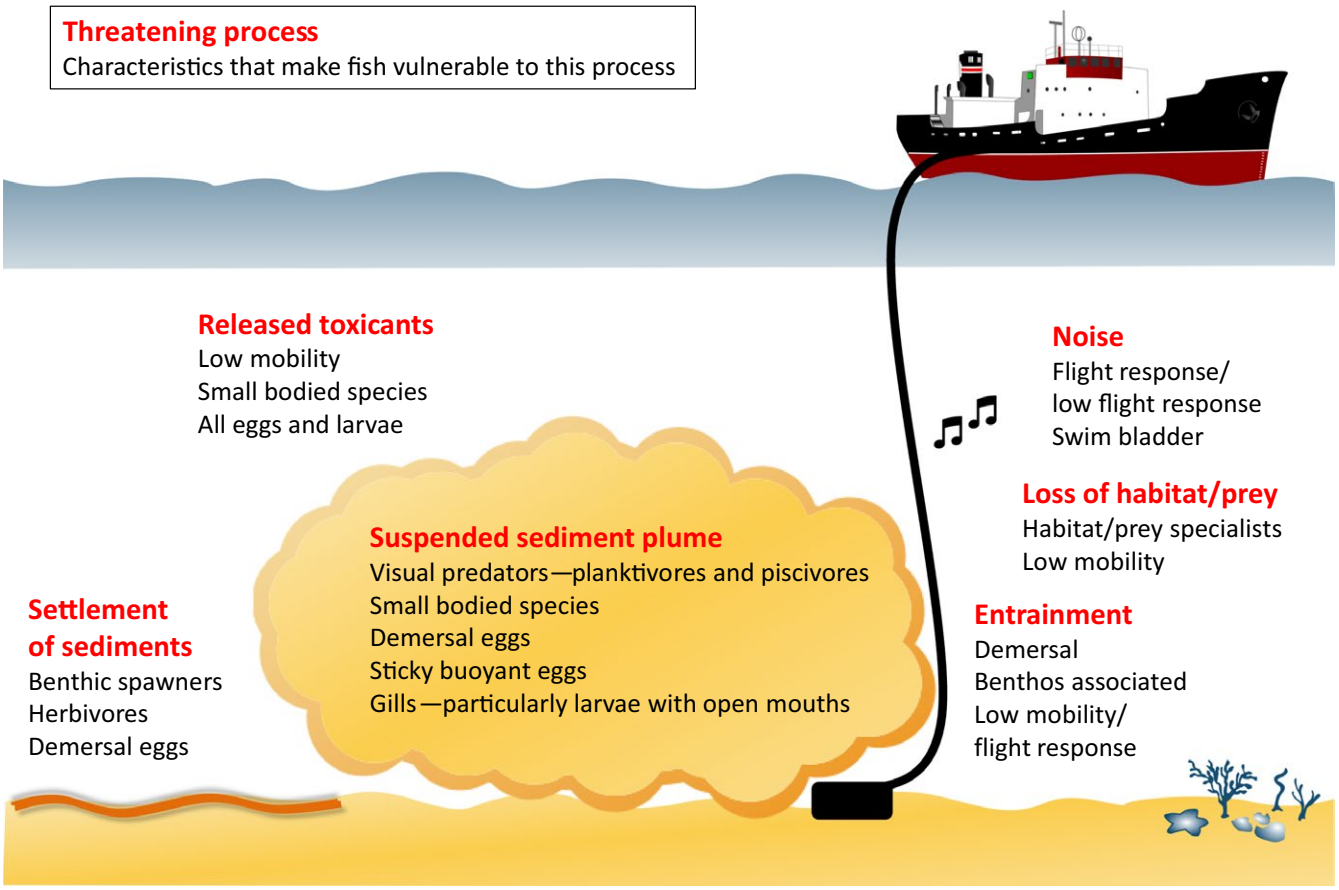

FIGURE 1 A schematic diagram of categories of potential effects of dredging on fish. [Colour figure can be viewed at wileyonlinelibrary.com]

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,

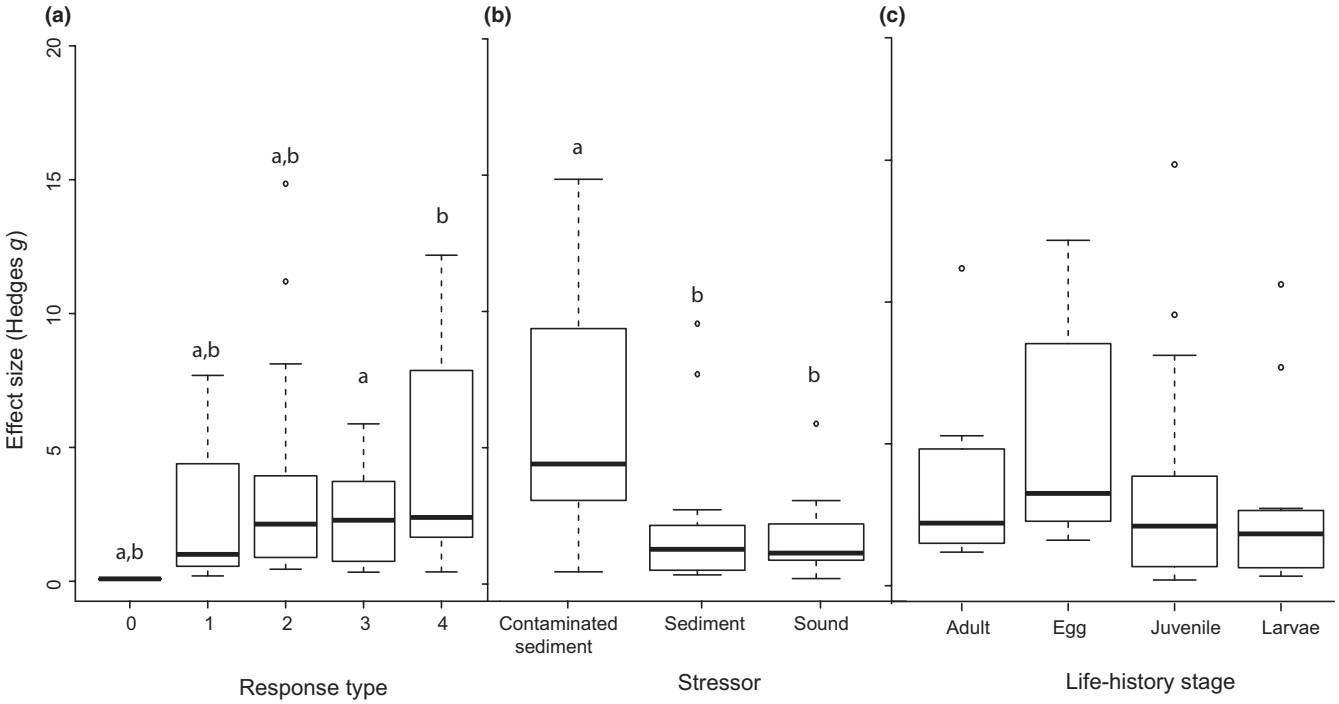

FIGURE 2 The impact of (a) response type, (b) stressor type, (c) life-history stage on effect size across all stressors. A response type of 0 = no effect, 1 = minor behavioural changes, 2 = minor physical damage or moderate behavioural changes, 3 = physiological impacts and 4 = increased mortality. Variables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

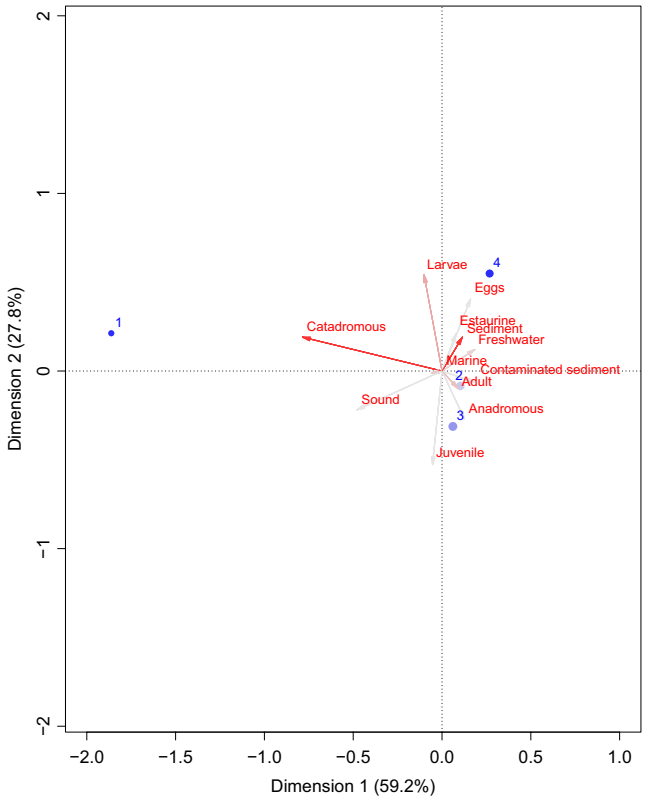

FIGURE 3 An asymmetric graph of the linear correspondence analysis, with the response type in the principal coordinates and the explanatory variables in reconstructions of the standardized residuals (square root of the relative frequency). Response type is represented by points, and the explanatory variables 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

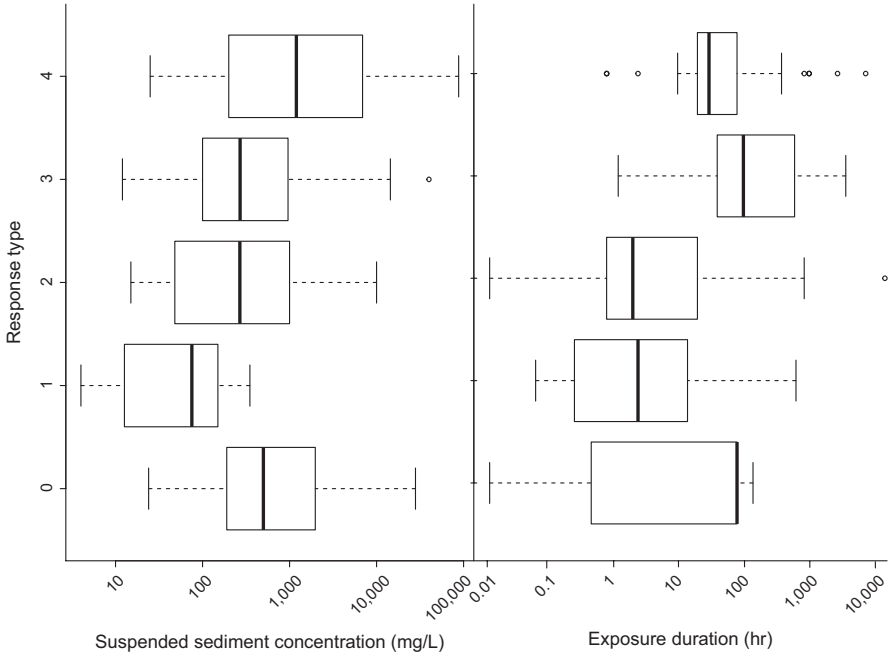

FIGURE 4 The impact of (a) suspended sediment concentration and (b) exposure duration on the type of effect elicited by suspended sediment. A response type of 0 = no effect, 1 = minor behavioural changes, 2 = minor physical damage or moderate behavioural changes, 3 = physiological impacts and 4 = increased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

3.1 Introduction and background

3.1.1 Management of dredge operations

Current management of dredge operations rely on a 'best practice' approach (EPA 2016) given that hydrodynamic models used to predict sediment plumes are often inaccurate and biological information is often limited or unavailable. In the planning stages, management may be based upon the avoidance of critical windows of environmental sensitivity (EPA 2016). During operation, most dredge projects rely on near real time monitoring of trigger values, which represent an upper limit in turbidity and sedimentation (PIANC 2010). If these values are exceeded, dredging operations are halted until values fall below these upper limits. Reliable trigger values, i.e. values that represent community sediment thresholds based on cause-effect relationships, need to be built on site specific information that takes into account local environmental conditions and community composition. Baseline surveys and continual monitoring are required to accurately set trigger values and enable managers to react if values are exceeded. This form of adaptive management relies on an understanding of both the physical and biological environment (Erftemeijer et al. 2012). Unfortunately, the associated costs and timeframes of implementing a comprehensive environmental impact assessment with continued monitoring often forces managers to apply trigger values developed from other sites (Nieuwaal 2001). These off-site trigger values may be either too low leading to elevated dredging costs, or too high resulting in increased damage to sensitive habitats (Erftemeijer et al. 2012).

Trigger values can be site specific and vary depending on distance away from the dredge site. Site specific trigger values are determined on whether or not they sit within the boundaries of one of the following three categories: the Zone of High Impact, the Zone of Moderate Impact, and the Zone of Influence (EPA 2016). The definition of these zones have been revised over the last few years, but typically relate to the amount of acceptable damage (e.g. total mortality in the zone of high impact) to key communities (EPA 2016). Within each of these zones, guidelines on the severity and duration of sediment events are provided for both coral and sea grass communities. These values are trigger values and if exceeded, dredge operations are suspended until sediment levels fall below recommended values. Note that no such trigger values are in place for fish communities. The size of these zones is highly dependent on site specific characteristics such as sediment types and hydrodynamics, and is often defined by hydrodynamic and sediment transport modelling prior to dredging activities. Typically the zone of high impact will extend <1 km from the dredge site whereas the zone of influence could extend 10's km. Models used to identify the various zones around dredge sites are also used to determine the most appropriate location for dredge spoils grounds. Yet, inadequate modelling in the past has led to incorrectly placed zones of influence and associated trigger values and, therefore, elevated damage to benthic communities (Pollock et al 2014).

To enable managers to accurately set reliable site specific trigger values, more biological information is required on a range of organism and community responses to sediments as well as improved hydrodynamic models. Field collection of these data is, however, difficult due to limited visibility often within high energy environments which impede diver access. Furthermore, diver access close to dredging sites is typically restricted due to health and safety concerns. Consequently, most assessments on organism responses to sediments are carried out in laboratories. Ensuring laboratory studies are representative of *in-situ* dredging conditions is critical, hence more information on sediment flux prior, during and following dredge operations is required to improve sediment transport models and allow researchers to re-create environmental realistic sediment exposure regimes during laboratory tests. These data can be collected using *in-situ* data loggers, but access to the site for data logger deployment and sediment flux data collected by dredging corporations is typically restricted. All available scientific and grey literature, to improve knowledge as well as identify knowledge gaps for targeted research is compiled by PIANC (PIANC 2010). Unfortunately, spatial variations in sediment sensitivity and general lack of biological information for most marine organisms make setting appropriate thresholds to sediments complicated. (Wenger et al. 2011, Erftemeijer et al. 2012). In studies that have examined the effects of suspended

sediment (SS) on different species, high variation in tolerance to SS has been observed. However, despite this variation, similar types of responses have been observed in species from all aquatic environments. The consistent nature of these observations indicates that it is currently possible to predict effects of dredging on fish, although the point at which these effects manifest is uncertain.

Temporal critical environmental windows are those periods of the year when the impacts of dredging and disposal activities on fish and other organisms have been deemed to be above the critical thresholds that allow dredging and disposal activities to be carried out without adversely affecting key marine organisms (EPA 2016, Reine et al. 1998, Suedel et al. 2008). For fish, critical environmental windows where dredging should be avoided, include important periods in their life history, such as those associated with reproduction and recruitment. Impacts of dredging on fishes may be significantly greater during these periods. Critical environmental windows may also have a spatial component, such as aggregation sites for mating and/or spawning. Recent research in Western Australia suggests that nearshore habitats such as macroalgal beds are critical in recruitment processes for many emperor (lethrinidae) species (Evans et al. 2014, Wilson et al. 2014). Knowledge of these time periods and locations is important to minimize the potential impacts of dredging on finfish communities. Most often, critical environmental windows are designed to provide opportunity for dredging whilst simultaneously protecting fish against the primary stressors generated by dredging and disposal operations. Such potential stressors on fish can include:

- hydraulic entrainment of fish eggs and larvae and juvenile fishes;
- increased suspended sediments and turbidity;
- resuspension of buried contaminated sediments which may release toxins and nutrients that can have acute or chronic effects on fish;
- loss of fish habitat through sedimentation, removal or degradation;
- increased noise; and
- Introduction of marine pests.

A significant amount of research has been conducted globally to try and determine the effect these stressors have on fish. In particular, there have been numerous studies that have examined the influence of increased suspended sediments and turbidity on single or groups of species of fish (e.g. Auld & Shubel 1978; Bash et al. 2001; Au et al. 2004; Wenger et al. 2012). A number of review papers have summarised the main findings of these studies over the years (e.g. Appelby & Scarratt 1989; Clarke & Wilber 2000; Wilber & Clarke 2001; Utne-Palm 2002; Bilotta & Brazier 2008; Kemp et al. 2011). Other potential stressors, such as those listed above, have also been studied for certain species, usually those of commercial importance.

Dredging operations can be stopped to avoid critical environmental windows for finfish, but at significant cost. It is essential that relevant critical ecological processes are identified and that appropriate levels of management intervention are put in place at the necessary spatial and temporal scales to avoid, or minimise potential impacts, but not un-necessarily constrain dredging activities.

Previous reviews on the effects of dredging have focussed on sedentary benthic fauna and flora (e.g. Bak 1978; Erftemeijer et al 2006; 2012). The effects on fish, which are of commercial, social, conservation and functional significance in aquatic systems, is yet to be critically reviewed. This is despite a growing body of literature that explicitly examines how threats associated with dredging can impact fish. In part 2 this review has synthesised the known effects of suspended sediment, hydraulic entrainment, and noise associated with dredging on fish. Part 3 examines the current state of knowledge on species of high value in Western Australian fisheries, or of known ecological or conservation importance to assess species most at risk to dredging. Finally, it provides recommendations on how this information can be used to minimize the environmental effects of dredging on Western Australian fish.

3.2 Framework for the environmental windows analysis

Critical environmental windows are periods of the year when fish are most susceptible to impacts due to the timing of a life history event (Suedel et al. 2008). Seasonal restrictions, periods when dredging operations are halted or scaled down, may be put in place during critical environmental windows. After reviewing the literature, we identified traits that make fish vulnerable to various stressors associated with dredging (Figure 1). We determined that eggs and larvae are the most vulnerable life history stages to dredging related stressors, and would therefore merit the implementation of seasonal restrictions. With this in mind, we undertook a review of the spawning times and pelagic larval duration of marine fish that are caught and retained by Western Australian commercial and recreational fishers. Literature was sourced from Google Scholar, Scopus and the ISI Web of Knowledge using the keywords: “fish”, “larvae”, “eggs”, “spawning” and “Western Australia”. We also used expert knowledge from the Western Australian Department of Fisheries to identify unpublished grey literature, reports and undocumented knowledge (See appendices 3, 4 and 5).

3.3 Results

3.3.1 Spawning periods in Western Australia

Of the 102 fish species targeted or retained by commercial and recreational fishers (appendix 4), spawning times were available for 60 (59%) species and the duration of spawning for 58 species (57%). The pelagic larval duration of only 17% of the species was available. Timing of recruitment was quantified for 11% of fish and only 7% of species had data on recruitment duration. Most notably, the 102 species we reviewed represent only a small portion of the total fish diversity of Western Australia and for the vast majority of these species very little information exists on their biology and ecology (S. Newman, unpublished data). For the 56 species where reproductive data was available, the peak spawning period was between October and April (appendix 1, Figure 2). However, up to 20% of the tropical and 30% of the temperate species spawned during the lowest period of spawning activity (July). There are also clear examples of latitudinal variation in peak spawning times particularly for species with large geographic ranges. For example, pink snapper (*Chrysophrys auratus*) spawn between April and October in the sub-tropical Gascoyne coast bioregion outside Shark Bay (latitude 26 °S). In the west coast bioregion around the Perth metropolitan area (32 °S), spawning occurs from August to January (Wakefield et al. 2015). Cockburn Sound, Warnbro Sound, and Owen anchorage are key spawning locations for this species, and pink snapper within Warnbro and Cockburn sounds are protected from fishing between October and January to protect spawning aggregations. Further south in the Albany region (35°S), spawning takes place over 2–3 months between September and December (Wakefield et al. 2015). Water temperature is a key trigger for spawning in this species, with spawning occurring at water temperatures of between 19°C and 21°C (Wakefield et al. 2015). The Shark Bay world heritage area in the Gascoyne coast bioregion is a unique location for this species. Shark Bay supports three different pink snapper stocks with high genetic diversity, and each is managed separately. Pink snapper in the eastern gulf spawn between May and July, and are protected from fishing during this time (Jackson et al. 2010). Within Denham Sound spawning also occurs between May and July. However, in the Freycinet estuary area spawning occurs between August and October (Jackson et al. 2010), and the area is closed to fishing from August 15th to September 30th. Patterns such as these highlight the need for location specific research into the reproductive biology of key fish species to inform when dredging is most appropriate. In some cases this information will need to be location specific. Given the significant overlap of spawning times between October and April, there is merit in considering the introduction of seasonal restrictions on dredging during this time, particularly for maintenance dredging, to mitigate effects Western Australian fish. This approach may be particularly relevant where risk analyses indicate potential impacts on fish species of high socio-economic or conservation value.

From the literature review and workshop we concluded that there were 12 traits which make fish vulnerable to dredging (appendix 4). These include:

1) Fish which are benthic spawners (settlement of sediment, hydraulic entrainment)
2) Fish which are pelagic spawners (suspended sediment)

3) Fish which have a long pelagic larval duration (PLD) (suspended sediment)
4) Demersal eggs (settlement of sediment, hydraulic entrainment)
5) Fish which are visual feeders (suspended sediment, light attenuation)
6) Fish which target highly mobile prey, particularly piscivores (suspended sediment, light attenuation)
7) Fish from clear water environments (suspended sediment)
8) Fish with specific habitat association (loss of habitat, suspended sediment, settlement of sediment, noise, hydraulic entrainment)
9) Fish which occupy vulnerable habitats that are restricted in coverage (loss of habitat)
10) Fish which have a small home range (suspended sediment, settlement of sediments noise, hydraulic entrainment)
11) Fish which are scraping herbivores (settlement of sediment, hydraulic entrainment)
12) Benthic dwelling fish (settlement of sediment, hydraulic entrainment)

3.3.2 Determining priorities for future research

Prioritising fish species

One of the key challenges is to prioritise the research that will aid managers to make decisions about the potential risks of dredging to fish given the large gaps in knowledge. Stelzenmuller et al. (2012) outline an approach to predict effects of dredging on fish. They developed an index which reflected species sensitivity to aggregate extraction based on approaches initially developed by Furness and Tasker (2000) and Garthe and Hüppop (2004). Stelzenmuller et al. (2012) described 'sensitivity' as the degree to which fish species respond to a pressure, and 'vulnerability' is the probability or likelihood that a component will be exposed to a pressure to which it is sensitive (Zacharias & Gregr, 2005). The index summed the scores of seven different factors (see Stelzenmuller et al. 2012 for more information) derived from species attributes which consider both their sensitivity and vulnerability. These attributes were deemed to be most responsive to impacts associated with aggregate extraction. Stelzenmuller et al. (2012) scored each of the seven factors below on a five-point scale from 1 (low) to 5 (high):

1) Geographical distribution: species with restricted distributions have the highest sensitivity score.
2) Threat status: determined using the IUCN redlist.
3) Importance to fisheries – economic importance to commercial fisheries (price × weight landed).
4) Habitat vulnerability: proportion of habitat vulnerable to dredging with information on habitat position and type and species usage of these habitats.
5) Ability to switch diet: examination of the species trophic guild and the impact of aggregate extraction on its prey.
6) Affinity to seabed: consideration given to species habitat and speed of movement.
7) Reproductive Strategy: position of eggs, position of post-larval stage and fecundity.

We have trialled and present a similar approach (appendix 2, Table 1). This priority assessment process uses a combination of characteristics of each of the species under consideration and combines both community values and vulnerability attributes. The sum of all the key attributes scores (i.e. economic significance, recreational significance, customary significance, social significance, IUCN/EPBC significance, reproductive strategy significance, recruitment significance, distribution significance, connectivity significance and habitat specificity significance) provides a rank rating across all of these values for each species under consideration. The objective of this sum of key attributes value is to provide an overall score across all attribute values. These values represent a combined total maximum score of 50. The objective of this sum of values is to rank and identify potential indicator species that are both valuable (fisheries, conservation, or ecological) and susceptible to dredging. The higher the combined score the greater the concern.

The process is a modification of the Ecosystem Based Management Framework proposed by Fletcher et al. (2010) and Fletcher (2015), and is an assessment of the characteristics of each species under consideration that combines both their commercial, recreational, customary and social values with vulnerability attributes.

The 13 species we assessed have a range of life history characteristics and community values. The silver cheeked toadfish (*Lagocephalus scelaratus*) is a common species with no commercial, recreational, customary or social values. It is a widespread species (Indo Pacific distribution) which a habitat generalist and is very resilient and not listed as being of concern for IUCN or the EPBC act. Hence, this species scores 8 out of 50. For demonstration purposes the species which ranked the highest sum score (30 out of 50) was the western blue groper (*Achoerodus gouldii*). This species has commercial, recreational, cultural and social significance and is listed as vulnerable due to its life history characteristics (long lived, late maturing, protogynous hermaphrodite). It is also a southern Australian endemic.

The assessment process highlights that the species of concern are those that are long lived with complex reproductive biology and limited distributions. Species of commercial and recreational significance tend to rank higher as their numbers are often reduced. Similarly, species listed as being threatened or vulnerable on the IUCN Red list or on the EBPC act have higher summed scores.

We have found this to be a useful process for identifying species that may be vulnerable to dredging activities, and so may be a priority for management. The process may also be used as an indicator of species where further dredging research resources should be targeted. This approach could be expanded to encompass a greater range of species, or to assess species that have been identified in a proposed development area during pre-development surveys. In such cases, this approach could be used to prioritise species for management purposes.

Research priorities

To make recommendations about the areas where further research should be prioritised we listed all of the key research questions identified during the workshop and review and then asked workshop participants to rank the gaps from the perspective of generating outcomes which would assist managers to minimising impacts to dredging and the potential threats to species of concern. Key research areas were identified around gaps in the knowledge about spawning and recruitment and the effects of dredging on fish (appendix 2,Table 3). There is a lack of basic ecological information for many Western Australian fish species (appendix 4) and the effects of suspended sediment, or vulnerability to dredging of most species is unknown (appendix 5). The prioritisation process (appendix 2, Table 1) can be used to guide the species or types of species towards which research should be directed to begin to fill these gaps.

3.3.3 Developing trigger values for Suspended Sediment Concentrations (SSCs)

Threshold reference values of sediment or contaminant levels considered detrimental to marine life are often used to trigger a management response when they are exceeded, such as halting or restricting dredging activities (Erftemeijer & Lewis 2006; Falkenberg & Styan 2014). Trigger values developed for management are derived from physical, chemical or biological threshold reference values. Site-specific physico-chemical trigger values for ecological health in Australia and New Zealand are generally based on the 80th percentile of recorded water quality data over 24 months at appropriate reference sites (ANZECC and ARMCANZ 2000). Where such data does not exist, trigger values are derived from modelling of continuous dose-response data that covers a sufficient range of species over a broad range of taxonomic groups (Chapman et al. 1996).

Due to a lack of quantitative information on specific risks associated with different dredging related pressures, threshold concentrations and exposure could only be examined for the effects of SS. However, information on guidelines for contaminant trigger values can be found in the Australian and New Zealand guidelines for fresh and marine water quality (ANZECC and ARMCANZ 2000). Information on the life history stage, the concentration of SS, and the exposure time were extracted from 71 studies, with the majority of studies (67%) testing juvenile and adult fish. Studies that measured SS using turbidity estimates were ultimately excluded, except when

conversion rates between turbidity and mg l^{-1} were known (appendix 2, Table 4). Since the grain size and angularity of sediment particles will have major implications for the potential effects, only studies with the same sediment type or grain size were compared.

The duration of exposure, measured endpoints and life-history stages in the reviewed studies were variable and reflect logistic constraints of laboratory exposures rather than realistic exposure scenarios. All studies reported continuous exposures lasting between 1.2 mins and 64 days (appendix 2, Table 4). The duration of daily dredging activities can theoretically vary from a few hours to 24 hours a day continuously (Nieuwaal 2001). Such variation in test parameters, life-stage and reported endpoints makes it difficult to collate results to provide reliable estimates for trigger values. The published data is such that it is not possible to include completely consistent datasets, so each reported value should be interpreted with caution until more consistent and locally relevant data can be incorporated into the models.

We examined the ability to derive no effect threshold values for fish exposed to very fine sediment with particles <4 µm using species sensitivity distributions (SSDs). Particle size distribution estimates from two dredging locations in the northwest of Western Australia indicate that up to 40-60% of sediments are less than or equal to 4 µm (GEMS 2012; Jones et al. 2016). SSDs fit empirically derived toxicity data to a cumulative probability distribution across taxonomic groups to allow the derivation of a concentration that will protect a defined percentage of species. All data points for a particular species are reduced to the one that demonstrates highest sensitivity. We used the Burrlioz 2.0 software (CSIRO 2015) to fit published data on the highest suspended sediment concentration that showed no effect on a species (NOEC) to a Burr Type III cumulative probability distribution. We report sediment concentrations to protect 25, 50, 80, 90, 95 and 99% of fish species. Confidence intervals (95%) were estimated using bootstrapping procedures.

There were 20 studies in the literature that fit our criteria, with both freshwater and marine species (appendix 2, Table 4). The studies were examining a range of endpoints, from behavioural changes to lethal changes (appendix 2, Table 4). The values chosen for the curve were the concentration where no effect of sediment was observed for the measured endpoint. This means that it is possible that other impacts from sediment may have occurred, but were not measured. Due to the limited number of studies available, all exposure times (ranging from 5 minutes to 96 hours) were included in the analysis (appendix 2, Table 4). The concentrations of very fine sediment that fish were able to withstand ranged from 9-500 mg L^{-1} (appendix 2, Table 4; appendix 1, Figure 3). Trigger values ranged from 166 mg L^{-1} to protect only 25% of species to 2.4 mg L^{-1} to protect 99% of species (appendix 2, Table 5; appendix 1, Figure 3). Although there is minimal published data on SS levels reached during dredging operations, Jones et al. (2015a) recorded several days of SS concentrations between 10-80 mg L^{-1} that extended up to 5 kilometres following a large scale capital dredging program in Western Australia (Fisher et al. 2015). Depending on the volume of sediment that must be excavated, the episodes of plume release and consequent settlement may persist for several days to weeks (Jones et al. 2015a) and affect areas up to 20 km from the dredge, though the most intense effects are typically within 5 km (Fisher et al. 2015). In such cases the trigger values derived here would be exceeded. The trigger value identified here should be considered cautiously due to the paucity of data, especially considering that the variance in values may be attributable to both differences in species and in life history stages tested. The estimates of the protection values are therefore likely to change if more data is added. This becomes important where specific species values are of interest and their sensitivity is unknown, particularly because the data used are not an unbiased cross section of species. It would be prudent that any future work done on the development of trigger values tests a range of species in order to aim towards ecosystem wide protection. Additionally, studies examining the effects of SS on visual acuity often do not take into consideration the light attenuation that would occur in the natural environment if fish were located 5-10 m below the surface (Jones et al. 2015a). Although literature on the effects of both light attenuation and suspended sediment shows mixed results (Miner & Stein 1993; Vogel & Beauchamp 1999), it is important to take this potential interacting effect into account when considering the development of trigger values.

In the case of dredging impacts where local fish communities need to be considered, it is unlikely that species of concern will have empirical toxicity data represented in the literature. Accordingly, the use of trigger values

derived solely from existing literature (as above) should be approached with caution. However, it is evident that decisions around protection levels need to be closely associated with what is being protected. Consequently, data on local species needs to be gathered and included in these datasets to improve the reliability of the trigger values. Alternatively, other management responses, such as species-specific tailoring of dredging times, could be considered.

3.4 Discussion

3.4.1 Critical Environmental Windows

Critical environmental windows may be used to manage the timing of dredge operations to minimise ecological impacts, though local information on life histories of species is required. Accordingly, the National Academies of Science Marine Board (2001) offered recommendations for infusing science into decisions regarding temporal restrictions of dredging projects. Our evaluation of the literature suggests early life history stages are most vulnerable to dredging, which may warrant consideration of a reduction in intensity, relocation, or the temporary closing down of dredge and spoil disposal operations during peak periods of spawning and recruitment. Although this review focuses on temporal environmental windows, spatial environmental windows could be equally important. Reduced or delayed dredging activities near recognized spawning aggregations or pathways between connected sites that are important for ontogenetic migrations would improve protection of species. Identification of key spatial environmental windows would also allow for dredging schedules in key areas to be implemented during certain times of the year where the risks would be minimal. There is spatial flexibility in the location of dredge spoil grounds. These should be located to minimise the resuspension of unconsolidated sediments, and to prevent any re-suspended sediments affecting critical habitats for spawning or recruitment downstream. The unconsolidated sediments in spoil grounds can be resuspended by wind and tidal forcing (Larcombe et al. 2001; Wolanski et al. 2008) or altered hydrodynamics as a result of development (PIANC 2010) and so can increase the possibility of interactions of re-suspended sediment with natural disturbances. For example, much of north-western Australia is affected by macro-tides, high levels of natural turbidity and cyclone disturbance. Sediment resuspension may interact with these natural disturbances in either an additive and / or synergistic fashion to impact critical habitats for fishes.

The information that is available for Western Australian fish species suggests that to limit the effect of dredging impacts on spawning to 50% of fish species or fewer, seasonal restrictions on dredging could be introduced during the peak spawning window between October and April. However, for most Western Australian fish species there was minimal information on timing or location of spawning and recruitment. This lack of information could lead both to unnecessary delays in dredging projects when actual risks are low or potentially large effects on fishes if impacts coincide with the timing of an unknown critical ecological process.

3.4.2 Developing trigger values for Suspended Sediment Concentrations (SSCs)

Existing studies on the effects of SS on fish allowed for the development of NOEC curves for very fine sediment only. Given the variability in exposure time, experimental design, and sediment type, these trigger values should be applied with caution. Improving the reliability of the trigger values will require local studies that evaluate impacts on resident species that are vulnerable to dredging and are of high social, economic and ecological value. The use of locally sourced sediment would also improve the reliability of the responses because sediment type can play a major role in the biological responses of organisms (Appelby & Scarratt 1989). However, studies of particle size distributions estimates from two dredging locations in the northwest of Western Australia found that up to 40-60% of sediment was less than 4 µm in size (GEMS 2012; Jones et al. 2016), suggesting that in the absence of additional information, the trigger values developed here could be implemented in Western Australia as a tool to guide when a management response is necessary. Additionally, previous studies have found that the sediment particles on coral reefs of most concern are the very fine-grained sediment (Bainbridge et al. 2012; Storlazzi et al. 2015). The trigger values developed here are useful in determining how fish in coral reef ecosystems are negatively affected by very fine grained suspended sediment.

If biological trigger values cannot be developed, it may be possible to develop physical trigger values, based on locally occurring SS concentrations (ANZECC and ARMCANZ 2000). However, the development of a turbidity

trigger value to initiate a management response is reliant on the ability to accurately measure SS. A frequently used model for compliance monitoring consists of collecting turbidity measurements at a specified distance down-current from the dredge and at specified depths in the water column. These measurements are then compared to similar measurements taken on the up-current side of the dredge. However, compliance monitoring adhering to this model has been shown to be arbitrary owing to the use of turbidity as a surrogate for more meaningful biological parameters (Clarke & Wilber 2008; Clarke et al. 2010). Additionally, turbidity is a proxy for light attenuation and while turbidity measurements for trigger values may be possible for species where light limitation is the main effect (Erftemeijer et al. 2005), it is not an appropriate metric for impacts on fish. Sediment can clog gills and reduce their efficiency (Appelby & Scarratt 1989). Larger and more angular sediment particles are also more likely to lodge between lamellae and cause physical damage to gill tissues and function (Bash et al. 2001; Servizi & Martens 1987). Furthermore, there is not a clear relationship between light attenuation, suspended sediment, and effects on fish (Collin & Hart 2015). Therefore, measurements of turbidity and light will not give a reliable indication of dredging effects on fish. Estimates of grain size and sediment concentration would increase confidence that local fish species could be better protected from dredging impacts

Although there have been several studies on the effects of SS on fish (e.g., Utne-Palm 2002; Wenger et al. 2015), there remain key knowledge gaps on the effects of different concentrations and exposure durations on different life history stages of fish, which must be addressed to ensure that trigger values that initiate management and timing of critical environmental windows are more robust. Given the large gaps in knowledge, a key challenge is prioritising research based on what will best aid decision makers in their efforts to manage potential impacts of dredging to fish. Though we did include criterion that categorise fish as herbivores or predators in prioritising effects of dredging on species, the relative importance of these roles among different species and locations was not considered. It would be important to improve our understanding of the ecological roles that species play. However, caution should be taken so as to avoid categorising a species as not ecologically important. Irrespective of the management strategy undertaken, research prior to a project allows for improved, site specific, understanding of both tolerance levels to SS and timing of vulnerable life history stages. Resources should be targeted to research focused both on understanding the effects of dredging related stressors on fish and on demographic and ecological information on species. Currently, the literature on dredging-related stressors is weighted toward examining the effects of suspended sediment, which was apparent in our inability to develop threshold reference values for other stressors. While suspended sediment is a ubiquitous stressor in any dredging project, further research needs to examine how contaminants released during dredging and noise associated with dredging can impact fish. Increased understanding of how each stressor acts alone or in combination will improve our ability to effectively manage potential impacts from dredging. The priority assessment process detailed above is a useful tool for identifying what species would be of concern for dredging activities and where further dredging related research resources should be targeted.

Ultimately, decisions about specific dredging project management practices will inherently involve compromises driven by uncertainties related to the probable scales of response and the probable effectiveness of a given management action in providing adequate protection. Although this report has primarily focused on two potential strategies, there are other management options pertaining to dredging operations that could be employed. For example, in cases where resuspension of contaminated sediment is of primary concern, effective management actions may include modifying the rate of output and mode of dredging, reducing overflow from hoppers, the application of silt curtains if the local conditions allow it, and adjusting the disposal method (Francingues & Palermo 2005, PIANC 2009). Decisions regarding implementation of such management practices should be made with full attention to factors that influence their performance.

3.5 Conclusion

This report demonstrates how our current understanding of the effects of dredging on fishes can be used to improve management practices. The studies reviewed to develop trigger values represent our current global understanding of the effects of SS on fish, which underscores just how difficult it is to take experimental studies from the literature and translate them into meaningful management actions. Out of several hundred studies that

exist in the literature regarding the effects of suspended sediment or turbidity on fish, we were only able to find 20 studies that contained enough suitable information to enable comparisons. Furthermore, based on the current state of information for the effects of other stressors on fish, it was impossible to develop trigger values for them. Similarly, the determination of life history stages that are most vulnerable to dredging was established through a thorough analysis of all known dredging related stressors and their effects on fish. Yet relevant life-history information existed for only a fraction the species in Western Australia. The identification of critical environmental windows to protect early life history stages of fish and locally relevant trigger values would increase confidence that local fish species could be protected from dredging impacts. Although we present a case study using Western Australian species, the vulnerable life history stages would be the same in any region, meaning a similar method could be applied where region specific information exists. There is a paucity of direct field measurements of the effects of dredging on fish, which needs to be rectified. Targeted research on the effects of local sediment types on fish and life-history characteristics is required. In addition to applying a sound scientific basis, managers must also consider socio-political pressures in developing policy and management strategies (Day 2008; Simpson et al. 2005). The priority assessment approach developed in this review combines the vulnerability of a species to dredging with the economic, cultural, and ecological consequences of dredging, thus ensuring that research can address the concerns of multiple stakeholders. Increased waterborne trade and the expansion of port facilities mean that dredging operations will continue to intensify globally (PIANC 2009). This report provides an important framework for determining how to best incorporate fish species and fisheries resources into dredging management plans, thus ensuring that dredging operations do not unnecessarily impact fish.

3.6 References

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4 Appendices

Appendix 1 - Figures (Parts 1, 2 and 3)

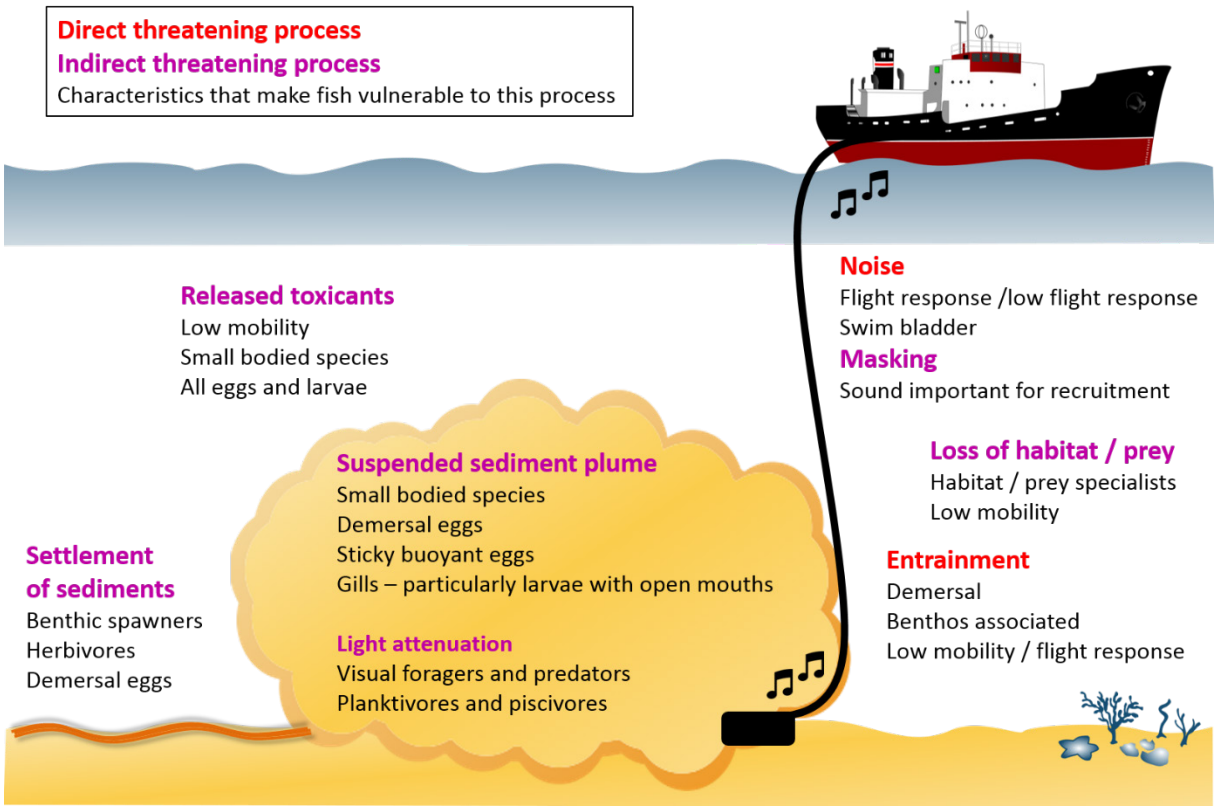

Figure 1 A schematic diagram of categories of potential effects of dredging on fish.

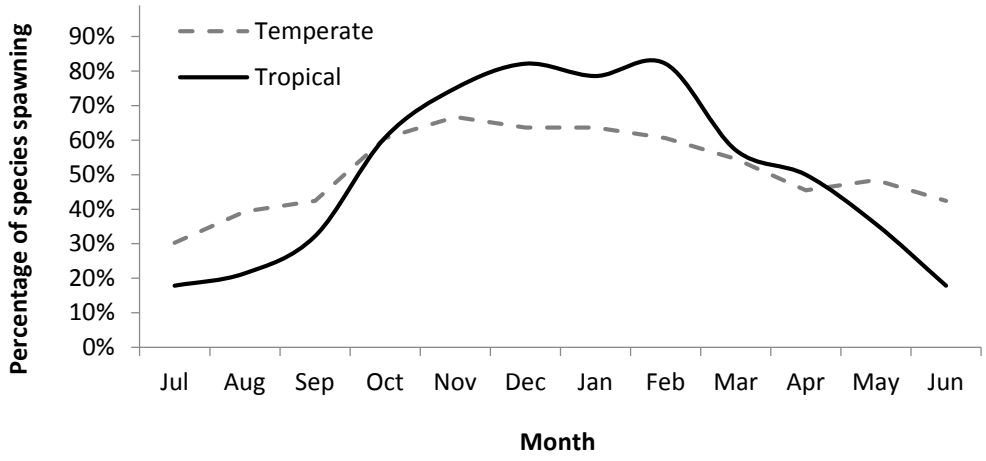

Figure 2 Known spawning periods for Western Australian fish species caught and retained by commercial and recreational fishers

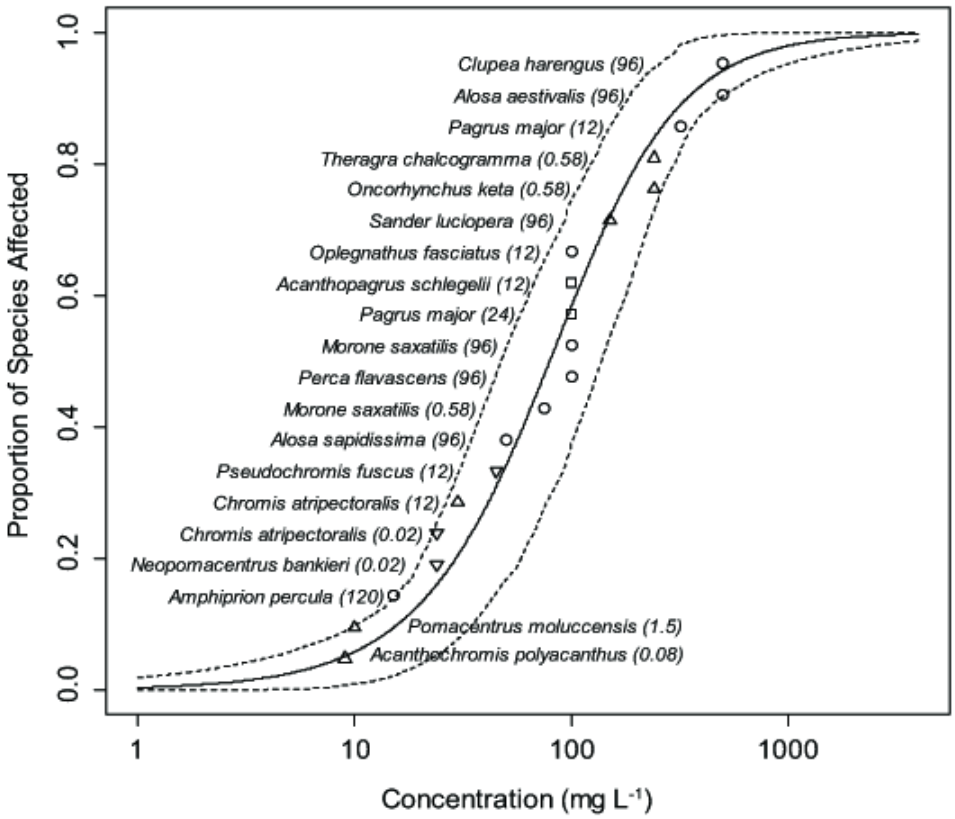

Figure 3 Burr Type III cumulative probability distribution for suspended sediment (<4 µm) concentrations (mg L-1) that impact fish (appendix 2, Table 5). Numbers in parentheses after species name are exposure times (hours). Dashed lines represent bootstrapped 95% confidence intervals. Fit and confidence intervals estimates were calculated by the Burrlioz 2.0 software (CSIRO 2015).

Appendix 2 - Tables (Parts 1, 2 and 3)

Table 1 A demonstration of the priority assessment process for 13 species. Where: E= Economic, R = Recreational, Cu = customary, S = Social, Co = IUCN/EPBC conservation, RS = Reproductive strategy, RE = Recruitment (year class resilience), D = Distribution/endemism, Conn. = Connectivity, HS = habitat specificity

Scientific name Common name	E	R	Cu	S	Co	RS	RE	D/E	Conn.	HS	Score
Lutjanus sebae Red emperor	4	5	1	3	0	4	2	2	1	2	**24**
Lutjanus russelli Saddletail Snapper	3	4	1	1	0	2	2	2	1	3	**19**
Epinephelus coioides Goldspotted Rockcod	3	4	1	1	3	2	2	2	1	4	**23**
Plectropomus leopardus Common Coral Trout	2	5	1	2	3	3	2	2	1	4	**25**
Choerodon rubescens Baldchin Groper	3	5	1	2	2	2	2	4	3	3	**27**
Glaucosoma hebraicum West Australian Dhufish	3	5	1	2	0	4	3	4	3	2	**27**
Achoerodus gouldii Western Blue Groper	2	4	2	3	4	4	2	3	3	3	**30**
Coris auricularis Western King Wrasse	0	2	0	0	0	4	2	4	3	2	**17**
Lagocephalus sceleratus Silver-cheeked toadfish	0	0	0	0	0	2	1	2	2	1	**8**
Carcharhinus obscurus Dusky Shark	3	1	0	2	4	4	5	2	1	1	**23**
Pristis zijsron Green sawfish	0	1	2	3	5	5	5	2	2	2	**27**
Hippocampus angustus Narrow-bellied Seahorse	2	0	0	3	4	5	2	4	3	4	**27**
Amphiprion rubrocinctus Red Anemonefish	2	1	0	3	0	4	2	3	3	5	**23**

Table 2 Full explanation of the codes used in Table 1

Code	Significance
E	Economic significance (E): What is the level of economic significance of this species to the wider community (i.e. does it contribute significantly to regional or State economies)?
R	Recreational Significance What is the relative priority of this species for recreational fishers within a bioregion, or, within distinct zones within a bioregion?
Cu	Customary Significance: What level of customary use or significance does the species under consideration have to indigenous communities of the area?
S	Social Significance (S): What is the level of cultural (social) concern or significance by the wider community for the species being considered (i.e. what consideration or value is placed on this species by non-recreational/non-commercial people)?
Co	IUCN/EPBC Significance: What is the IUCN and/or EPBC assessment rating of each species based on the most recent assessment (see IUCN red list for rankings)?
RS	Reproductive Strategy Significance: Is the species under consideration a broadcast spawner year round or are there discreet concise spawning windows?
RE	Recruitment (year class resilience) Is recruitment constant and consistent or is it highly variable and episodic?
D	Distribution/Endemism Significance: Is the species under consideration widely distributed or is it endemic to a region/location?
Conn.	Connectivity Significance: What is the stock or population structure of the species under consideration?
HS	Habitat specificity: What are the habitat requirements for the species under consideration, are they restricted and/or specific or do they occupy multiple habitats?

Table 3 Priority research areas as identified during the 2013 workshop

Research area or question	Priority
Spawning	
What are the spawning times, duration and spawning locations of species of concern?	High
What is the spawning behaviour of species of concern (i.e. single schools at one location)?	High
What is the pelagic larval duration for species of concern?	Medium
What are the environmental cues/conditions that result in spawning?	Medium
Where do the larvae of species of concern spend their pelagic period?	Medium
Recruitment	
When does recruitment occur for species of concern and into what habitats?	High
Is the timing of recruitment the same across the entire range of a species	High
What is the availability (area) of essential habitat throughout the recruitment range?	High
Is there an ontogenetic shift in habitat use for species of concern?	Medium
Effects of dredging	
What are the effects of sediment and noise on the larvae, recruits and adult stage of species of concern?	High
Will increased sediment and turbidity affect the ability for fish to use olfactory and visual cues to locate settlement habitat (and conspecifics, and avoid predators)?	Medium
Are fishes food resources negatively affected by dredging activities and increased sediment?	Medium
How does dredging affect respiration and thus body condition?	Medium
Does dredging effect primary productivity, and what are the flow-on effects for fish assemblages?	Medium
For benthic spawners does an increased sediment load interfere with the attachment of benthic eggs?	Medium
Are there physiological or behavioural adaptations to turbid environments?	Medium
Does dredging affect reproductive development– e.g. delayed onset of spawning?	Medium
Does dredging affect fertilisation rates and total reproductive output?	Medium
Do the acoustics of coastal dredging activities affect the navigational abilities of larvae and recruits?	Low
How does dredging affect different stages of larval life as swimming ability increases (e.g. pre and post-flexion)?	Low
Are fish's ability to detect, distinguish and capture pelagic food items compromised by dredging?	Low
Does increased turbidity from dredging affect the outcomes of predator / prey interactions?	Low
Does turbidity affect foraging distances?	Low
Does turbidity interfere with mating and courtship?	Low
Are important settlement microhabitats affected by increased sediment fallout through infilling or lethal / sub-lethal effects on biotic habitats?	Low
Does dredging affect levels of disease in fishes at any life history stage?	Low
Are some functional groups more susceptible to levels of disease due to direct impacts on feeding?	Low
Does dredging affect levels of fish parasitism and disease?	Low
Does dredging affect other aspects of development in fishes? Is there evidence that benthic spawners may provision their eggs or egg casings differently in high sediment environments?	Low

Table 4 Studies used to derive NOEC trigger values for very fine-grained (<4µm) sediment. All concentrations below are the maximum concentration in each study that did not affect the species. L = Larvae, J = Juvenile, A = Adult, E = Eggs

SSC ($g\ L^{-1}$)	Species Family	Common name	Life stage	Exposure time (hours)	Sediment type	Source
50	*Alosa sapidissima* Clupeidae	American shad	L	96	natural	Auld & Schubel 1978
100	*Perca flavescens*, Percidae	Yellow perch	L	96	natural	Auld & Schubel 1978
100	*Morone saxatilis*, Moronidae	Striped bass	L	96	natural	Auld & Schubel 1978
500	*Alosa aestivalis*, Clupeidae	Blueback herring	L	96	natural	Auld & Schubel 1978
500	*Clupea harengus*, Clupeidae	Atlantic herring	L	96	natural	Auld & Schubel 1978
75	*Morone saxatilis* Moronidae	Striped bass	L	0.58	Kaolin	Breitburg 1988
240	*Oncorhynchus keta* Salmonidae	Chum salmon	J	0.58	bentonite	de Robertis et al. 2003
240	*Theragra chalcogramma* Gadidae	Walleye pollock	J	0.58	bentonite	de Robertis et al. 2003
15	*Amphiprion percula* Pomacentridae	Orange clownfish	L	120	bentonite	Hess et al. 2015
100	*Pagrus major* Sparidae	Seabream	E	24	Kaolin	Isono et al. 1998
100	*Acanthopagrus schlegelii*, Sparidae	Blackhead seabream	E	24	Kaolin	Isono et al. 1998
100	*Oplegnathus fasciatus*, Oplegnathidae	Striped beakfish	L	12	Kaolin	Isono et al. 1998
320	*Pagrus major* Sparidae	Seabream	L	12	Kaolin	Isono et al. 1998
24	*Neopomacentrus bankieri* Pomacentridae	Chinese demoiselle	A	0.2	bentonite	Johansen & Jones 2013
24	*Chromis atripectoralis* Pomacentridae	Black-axil chromis	A	0.2	bentonite	Johansen & Jones 2013
9	*Acanthochromis polyacanthus* Pomacentridae	Spiny chromis	J	0.08	Kaolin	Leahy et al. 2011
150	*Sander lucioperca* Percidae	Pike-perch	J	96	bentonite	Ljunggren & Sandstrom 2007
10	*Pomacentrus moluccensis* Pomacentridae	Lemon damsel	J	1.5	bentonite	Wenger & McCormick 2013
45	*Pseudochromis fuscus* Pseudochromidae	Dottyback	A	12	bentonite	Wenger et al. 2013
30	*Chromis atripectoralis* Pomacentridae	Black-axil chromis	J	12	bentonite	Wenger et al. 2013

Table 5 Sediment stress threshold values for the protection of different proportions of species. Values are derived from a Burr Type III cumulative probability distribution fitted to NOEC data for impact on fishes (table 1) and confidence intervals (CIs) are bootstrapped 95% CIs. Values were fitted and CIs were estimated by the Burrlioz 2.0 software (CSIRO 2015).

Species Protected	Lower CI	Protection Value ($mg\ L^{-1}$)	Upper CI
25%	99	166	269
50%	47	79	147
80%	20	29	59
90%	10	16	35
95%	4.1	8.8	20.2
99%	0.4	2.4	10.7

Box 2: Relationships between suspended sediment and coral reef fishes in northwest Australia

A recent study conducted by the University of Western Australia and the WA Department of Parks and Wildlife used Underwater visual census, baited remote underwater stereo-video, and diver operated stereo-video to sample the fish assemblage and quantify habitat composition and complexity at 16 sites across a cross-shelf turbidity gradient in the nearshore Pilbara region of Western Australia. The study area experiences varying natural turbidity levels, and included some areas that had been recently impacted by dredging programs in the Pilbara region. The suspended sediment characters used in the models were long-term (12 year) remotely sensed total suspended solids (TSS), short term (three year) remotely sensed TSS, and in situ turbidity (NTU).

Increasing suspended sediment resulted in decreased species richness of fishes. However, no change in the total number of individual fish was found across the turbidity gradient. Negative relationships were found between suspended sediment and the abundance of both herbivorous scrapers and planktivorous omnivores. This reduction in abundance of planktivorous omnivores with increasing turbidity could be attributed to declining visual foraging success. The abundance of herbivorous scrapers was most strongly influenced by habitat rugosity, but also showed a negative response to increasing suspended sediment. This pattern may be due to increased sediment affecting the epilithic algal matrix, and thus decreasing the quality of food available for herbivorous scrapers. However, the relationship between suspended sediment and herbivorous scrapers is complex, as their abundance increases with decreasing live coral cover. So as increasing suspended sediment may drive a decrease in live coral cover, and an associated increase in the abundance of turf algae, the abundance of herbivorous scrapers may in turn increase.

The functional groupings of planktivorous omnivorous and scraping herbivorous fishes are vulnerable to elevated suspended sediment loads. Species from within these functional groups could be considered for selection as indicator species when designing management plans for anthropogenic activities that will influence the suspended sediment regime of the Pilbara region.

Reference

Moustaka M (2016) The negative influence of suspended sediment on a fish assemblage in northwest Australia. BSc Honours research dissertation, The University of Western Australia, Crawley, Western Australia, November 2016

Acknowledgements

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Appendix 3

Definitions of the scores assigned during the priority assessment process (appendix 2, Table 1)

E	**Economic significance (E): What is the level of economic significance of this species to the wider community (i.e. does it contribute significantly to regional or State economies)?**
0	Not of interest, no value
1	Minimal value or contribution to the economy
2	Some value or contribution to the economy (<5% of total catch)
3	Moderate value or contribution to the economy
4	A major species that contributes to regional and/or state economies
5	A dominant species that contributes significantly to regional and/or state economies
R	**Recreational Significance What is the relative priority of this species for recreational fishers within a bioregion, or, within distinct zones within a bioregion?**
0	Not relevant (i.e. NOT a recreational species)
1	Small incidental take only
2	Only a secondary species (e.g. flathead)
3	Highly targeted species, but only by a few (e.g. billfish)
4	A popularly caught species but not one of the primary species
5	One of the primary target species in the bioregion/zone (e.g. snapper, dhufish, tailor, herring)
Cu	**Customary Significance: What level of customary use or significance does the species under consideration have to indigenous communities of the area?**
0	None or virtually no customary significance
1	Minimal
2	Some
3	Moderate
4	A major species for customary use
5	High level of use, or of primary significance
S	**Social Significance (S): What is the level of cultural (social) concern or significance by the wider community for the species being considered (i.e. what consideration or value is placed on this species by non-recreational/non-commercial people)?**
0	Not relevant, no value
1	Minimal additional social value
2	Some broader community issues involved
3	Identifiable community concern issues
4	Issue is causing major troubles in region (e.g. Swan River fish kills)
5	Statewide issue of public concern (e.g. catching dolphins/whales)
Co	**IUCN/EPBC Significance: What is the IUCN and/or EPBC assessment rating of each species based on the most recent assessment (see IUCN red list for rankings)?**
0	Not evaluated.
1	Data deficient
2	Least concern
3	Near threatened
4	Vulnerable
5	Endangered

RS	**Reproductive Strategy Significance: Is the species under consideration a broadcast spawner year round or are there discreet concise spawning windows?**
1	Broadcast spawning in small groups throughout distribution, or year round spawning with high fecundity
2	Spawning period 4-8 months
3	Spawning period 3-4 months
4	Complex hierarchy (sex change & high atresia midseason), relatively small GSI of male and/or female (e.g. G. hebraicum), pair spawning or spawning period 1-2 months
5	Spawning aggregations every > 500 km (e.g. P. auratus, A. japonicus), short spawning period < 30 d, high parental care, low fecundity (e.g. C. macrocephalus)
RE	**Recruitment (year class resilience) Significance: Is recruitment constant and consistent or is it highly variable and episodic?**
1	Year round (relatively consistent)
2	Restricted annually with wide broadcast distribution of progeny
3	Subject to equal periods of poor and strong recruitment (e.g. G. hebraicum)
4	Poor with few occasional strong years (e.g. 2 per 10 yr, P. auratus)
5	Exceptionally low fecundity (e.g. many elasmobranchs)
D	**Distribution/Endemism Significance: Is the species under consideration widely distributed or is it endemic to a region/location?**
1	Circum-global
2	Indo-Pacific
3	Endemic Australia
4	Endemic Western Australia
5	Strong habitat association during one or many life stages with such habitat limited in WA (e.g. *C. macrocephalus*)
Conn	**Connectivity Significance: What is the stock or population structure of the species under consideration?**
1	One population over many Bioregions
2	Multiple populations throughout Indo-Pacific
3	One population with restricted range within Western Australia or Australia (endemic)
4	Multiple populations with restricted range within Western Australia (endemic)
5	Multiple discreet populations within a single Bioregion
Hs	**Habitat specificity: What are the habitat requirements for the species under consideration, are they restricted and/or specific or do they occupy multiple habitats?**
1	A habitat generalist that can occupy multiple habitats and/or is pelagic
2	Moderate range of habitat requirements
3	Limited habitat plasticity – e.g. can occupy various reef-type habitats only
4	Somewhat restricted to specific habitats – e.g. estuarine/freshwater only
5	Very restricted habitats and/or habitat requirements (e.g. *Amphiprion spp*.)

Appendix 4

Status of knowledge for spawning and recruitment for Western Australian marine fishes targeted or retained by commercial and recreational fishers Abbreviations Aquatic zone: Es = Estuarine, Ns = Nearshore, In= Inshore, Os = Offshore, P = Pelagic. Abbreviations Bioregions: NCB = North Coast Bioregion, GCB = Gascoyne Coast Bioregion, WCB = West Coast Bioregion, SCB = South Coast Bioregion, U = Unknown

	Spawning					**Recruitment**				**Bioregions**	**References**
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Arripidae											
Arripis georgianus Australian Herring	Apr-Jun (peak late May/early Jun)	3 months	West Coast Bioregion only. Reef, sand, weed	Variable between regions- increases with distance from spawning area	Es, Ns	Jun-Sep (variable between regions- increases with distance from spawning area)	4 months- Variable between regions- increases with distance from spawning area	Shallow nurseries with vegetation	Ns	GCB, WCB, SCB	1
A.truttaceus Western Australian Salmon	Feb-Jun (peak Apr/May)	5 months	Southern part of West Coast Bioregion	u	Ns	u	u	Soft substrate, shallow sheltered bays	Es, Ns	WCB, SCB	2
Berycidae											
Centroberyx gerrardi Bight Redfish	Jan-Apr	4 months	Aggregations, Reef	u	In	u	u	u	u	WCB, SCB	Preliminary data
C. australis Yelloweye Redfish	u	u	u	u	u	u	u	u	u	GCB, WCB, SCB	
C. lineatus	u	u	u	In	u	u	u	u	u	GCB, WCB, SCB	

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Swallowtail											
Carangidae											
Pseudocaranx sp. Silver Trevally	Jul-Dec (peak Sep- Dec)	6 months	Reef	u	Ns, In	u	u	u	Es, Ns	GCB?, WCB, SCB	3
Seriola hippos Samsonfish	Sept-Mar	7 months	Aggregations, Structure	u	In	u	u	Surface, underneath floating structure (detached seagrass, algae etc..)	Ns, In, P	GCB?, WCB, SCB	4
S. dumerili Amberjack	u	u	u	u	u	u	u	u	u	NCB, GCB, WCB?	
Caranx ignobilis Giant Trevally	u	u	u	u	u	u	u	u	u	NCB, GCB, WCB	
Gnathanodon speciosus Golden Trevally	u	u	u	u	u	u	u	u	u	NCB, GCB	
Carangoides gymnostethus	u	u	u	u	u	u	u	u	u	NCB, GCB	

	Spawning					Recruitment				Bioregions	References
	Time of year	Duration	Habitat	Pelagic larval or propagule duration	Aquatic zone	Time of year	Duration	Habitat	Aquatic zone		
Bludger Trevally											
C. malabaricus	u	u	u	u	u	u	u	u	u	NCB, GCB, WCB, SCB	
Malabar Trevally											
Centrolophidae											
Hyperoglyphe antarctica	Feb-Mar	2 months	Aggregations, Reef	u	In, Os	u	u	u	u	GCB?, WCB, SCB	Preliminary data
Blue-eye Trevalla											
Centropomidae											
Lates calcarifer	u	u	u	u	Es, Ns	u	u	u	u	NCB, GCB	
Barramundi											
Cheilodactylidae											
Nemadactylus valenciennesi	Nov-Jun	8 months	Reef, Lower West Coast only	u	Ns, In	u	u	Predominantly South Coast	Ns, In	WCB, SCB	5
Blue Morwong											
Glaucosomatidae											

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Glaucosoma buergeri	u	u	Reef	u	Ns, In	u	u	u	u	NCB, GCB	
Northern Pearl Perch											
Glaucosoma hebraicum	Oct-May (peak Dec-Mar)	8 months	Pairs, Social, Reef	u	In	u	u	Sand inundated reef, Sponge, Seagrass	Ns, In	GCB, WCB, SCB	6
West Australian Dhufish											
Haemulidae											
Plectorhinchus flavomaculatus	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB, WCB	
Goldspotted Sweetlips											
Diagramma labiosum	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB, WCB	
Painted Sweetlips											
Labridae											
Achoerodus gouldii	Jun-Oct	5 months	Pairs, Social, Reef	u	In	u	u	u	Ns	WCB, SCB	7
Western Blue Groper											
Bodianus frenchii	Oct-Feb	5 months	Pairs, Social, Reef	u	Ns, In	u	u	u	u	WCB, SCB	8

	Spawning					Recruitment				Bioregions	References
	Time of year	Duration	Habitat	Pelagic larval or propagule duration	Aquatic zone	Time of year	Duration	Habitat	Aquatic zone		
Foxfish											
Choerodon rubescens Baldchin Groper	Jul-Jan (peak Sep-Jan)	7 months	Pairs, Social, Reef	u	Ns, In	u	u	u	u	GCB, WCB, SCB	9, 10
C. schoenleinii Blackspot Tuskfish	Sept-Dec	4 months	Pairs, Social, Reef	u	Ns, In	u	u	u	u	NCB, GCB, WCB?	9
C. cauteroma Bluespotted Tuskfish	Apr-Dec	9 months	Pairs, Social, Reef	u	Ns, In	u	u	u	u	NCB, GCB, WCB?	9
C. cyanodus Blue Tuskfish	Nov-Feb	4 months	Pairs, Social, Reef	u	Ns, In	u	u	u	u	NCB, GCB, WCB?	9
C. cephalotes Purple Tuskfish	u	u	Pairs, Social, Reef	u	Ns, In	u	u	u	u	NCB, GCB, WCB?	Preliminary data
Coris auricularis Western King Wrasse	Apr-Jun	3 months	Pairs, Social, Reef	u	Ns, In	u	u	u	u	WCB, SCB	11, 12
Notolabrus parilus	Jul-Oct	4 months	Pairs, Social, Reef	u	Ns, In	u	u	u	u	WCB, SCB	11, 12

	Spawning					Recruitment				Bioregions	References
	Time of year	Duration	Habitat	Pelagic larval or propagule duration	Aquatic zone	Time of year	Duration	Habitat	Aquatic zone		
Brownspotted Wrasse											
Ophthalmolepis lineolatus Maori Wrasse	Sept-Feb	6 months	Pairs, Social, Reef	u	Ns, In	u	u	u	u	WCB, SCB	11, 12
Lethrinidae											
Gymnocranius grandoculis Robinson's Seabream	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB, WCB?	
Lethrinus laticaudis Grass Emperor	Dec-Mar	4 months	Aggregations, reef	u	Ns, In	u	u	u	u	NCB, GCB	13
L. miniatus Redthroat Emperor	Oct-Feb	5 months	u	u	Ns, In	u	u	u	u	NCB, GCB, WCB	Preliminary data
L. nebulosus Spangled Emperor	Oct-May (peak Nov-Mar)	8 months	Aggregations, reef	u	Ns, In	u	u	u	u	NCB, GCB, WCB	14, 15
L. olivaceus	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB, WCB	

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Longnose Emperor											
L. punctulatus	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB	
Bluespotted Emperor											
Lutjanidae											
Lutjanus argentimaculatus	Oct-Mar	6 months	Aggregations, Os reefs	u	In	u	u	Freshwater, brackish and Estuarine	Es,Ns	NCB, GCB, WCB	16
Mangrove Jack											
L. carponotatus	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB, WCB	
Stripey Snapper											
L. erythropterus	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB	
Crimson Snapper											
L. johnii	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB?	
Golden Snapper											
L.lemniscatus	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB	
Darktail Snapper											

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
L. malabaricus	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB, WCB?	
Saddletail Snapper											
L. russelli	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB, WCB?	
Moses Snapper											
L. sebae	Jan-Dec (peak Oct-Mar)	12 months	u	u	In	u	u	u	u	NCB, GCB, WCB	17
Red Emperor											
L. vitta	u	u	u	u	Ns, In	u	u	u	u	NCB, GCB	
Brownstripe Snapper											
Pristipomoides multidens	Jan-Apr	4 months	Aggregations	u	In	u	u	u	u	NCB, GCB	18
Goldband Snapper											
P. typus	u	u	u	u	In	u	u	u	u	NCB	
Sharptooth Snapper											
Etelis carbunculus	Nov-Feb	4 months	u	u	In, Os	u	u	u	u	NCB, GCB, WCB?	Preliminary data
Ruby Snapper											

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
E. coruscans Flame Snapper	u	u	u	u	In, Os	u	u	u	u	NCB, GCB	
Symphorus nematophorus Chinamanfish	u	u	u	u	In	u	u	u	u	NCB, GCB, WCB	
Polynemidae											
Eleutheronema tetradactylum Blue Threadfin	Aug-Jan	6 months	Sand/Mud beach habitat	u	Es,Ns	u	u	Sand/Mud beach habitat	Es,Ns	NCB, GCB?	19
Polydactylus macrochir King Threadfin	Sept-Feb	6 months	Sand/Mud beach habitat	u	Es,Ns	u	u	Sand/Mud beach habitat	Es,Ns	NCB	20
Polyprionidae											
Polyprion oxygeneois Hapuku	May-Sept	5 months	u	u	In, Os	u	u	Floating debris	P	WCB, SCB	21
Polyprion americanus	Mar-Jun	4 months	u	u	Os	u	u	Floating debris	P	WCB, SCB	22

	Spawning					Recruitment				Bioregions	References
	Time of year	Duration	Habitat	Pelagic larval or propagule duration	Aquatic zone	Time of year	Duration	Habitat	Aquatic zone		
Bass Groper											
Platycephalidae											
Platycephalus endrachtensis	Oct-May (peak Nov-Mar)	8 months	u	u	Es	u	u	u	Es	NCB, GCB, WCB, SCB	23
Yellowtail Flathead (Swan River Estuary)											
Sciaenidae											
Argyrosomus japonicas	Oct-May (peak Nov-Jan)	8 months	Night, Reef, Aggregations	u	Es, Ns,In	u	u	u	Es, Ns	NCB? GCB, WCB, SCB	24
Mulloway											
Protonibea diacanthus	u	u	u	u	Es. Ns	u	u	u	Es, Ns	NCB, GCB	
Black Jewfish											
Serranidae											
Epinephelides armatus	Oct-Jun	9 months	Reef	u	Ns, In	u	u	u	u	GCB, WCB, SCB	25
Breaksea Cod											
Othos dentex	Sep-Mar (peak Nov-Jan)	7 months	Pairs, Reef	u	Ns, In	u	u	u	u	WCB, SCB	26

	Spawning					Recruitment				Bioregions	References
	Time of year	Duration	Habitat	Pelagic larval or propagule duration	Aquatic zone	Time of year	Duration	Habitat	Aquatic zone		
Harlequin Fish											
Epinephelidae											
Epinephelus areolatus	u	u	u	u	u	u	u	u	u	NCB, GCB	
Yellowspotted Rockcod											
E. bilobatus	u	u	u	u	u	u	u	u	u	NCB, GCB	
Frostback Rockcod											
E. bleekeri	u	u	u	u	u	u	u	u	u	NCB	
Duskytail Grouper											
E. coioides	Jan-Dec (peak Oct-Mar)	12 months	Os Reefs, Pairs/social	U	In	u	u	Shallow, mangrove	Es, Ns	NCB, GCB, WCB	27
Goldspotted Rockcod											
E. malabaricus	Jan-Dec (peak Oct-Mar)	12 months	Os Reefs, Pairs/social	u	In	u	u	Shallow, mangrove	Es, Ns	NCB, GCB, WCB?	27
Blackspotted Rockcod											
E. morrhua	u	u	u	u	In	u	u	u	u	NCB	
Comet Grouper											

	Spawning					Recruitment				Bioregions	References
	Time of year	Duration	Habitat	Pelagic larval or propagule duration	Aquatic zone	Time of year	Duration	Habitat	Aquatic zone		
E. multinotatus	u	u	u	u	In	u	u	u	u	NCB, GCB, WCB	
Rankin Cod											
E. radiatus	u	u	u	u	In	u	u	u	u	NCB, GCB	
Radiant Rockcod											
E. rivulatus	Jul-Dec	6 months	Pairs/social	u	Ns, In	u	u	u	u	NCB, GCB, WCB	28
Chinaman Rockcod											
Plectropomus areolatus	u	u	u	u	In	u	u	u	u	NCB, GCB	
Passionfruit Coral trout											
P. leopardus	Dec-Feb	3 months	u	u	u	u	u	u	u	NCB, GCB, WCB	Preliminary data
Common Coral Trout											
P. maculatus	u	u	u	u	u	u	u	u	u	NCB, GCB, WCB	
Barcheek Coral Trout											
Hyporthodus octofasciatus	Oct-Feb	5 months	u	u	In. Os	u	u	u	u	NCB, GCB, WCB?	29
Eightbar Grouper											

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Sillaginidae											
Sillaginodes punctata	May-Aug (peak Jun-Jul)	4 months	Reef, sand, weed	u	Ns, In (to 50m)	Sep-Apr	8 months	patchy sand/seagrass	Es, Ns	WCB, SCB	30, 31
King George Whiting											
Sillago burrus	Nov-May	7 months	Sand, silt, mud	u	Ns	u	u	u	Es, Ns	NCB, GCB, WCB?	31, 32
Western Trumpeter Whiting											
S. vittata	Oct-May	8 months	Sand	u	Ns	u	u	Sand	Ns	GCB, WCB	31, 32
Western School Whiting											
S. cf. robusta	Dec-Aug	9 months	u	u	Ns, In	u	u	u	Ns, In	GCB, WCB, SCB?	31, 33
Stout Whiting											
S. bassensis	Jan-Dec (peak Dec-Mar)	12 months	Sand	u	In (20-35m)	All year (peak Nov-Apr)	12 months	u	Ns	WCB, SCB	31, 33
Southern School Whiting											
S. schomburgkii	Aug-Mar	8 months	Sand	u	Es, Ns	u	u	Sand	Es, Ns	GCB, WCB, SCB	34
Yellowfin Whiting (Shark Bay)											
S. schomburgkii	Oct-Mar (peak Dec-Feb)	6 months	Sand	u	Es, Ns	u	u	Sand	Es, Ns	GCB, WCB, SCB	35

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Yellowfin Whiting (West Coast Bioregion)											
S. lutea Mud Whiting	u	u	Mud, silt	u	Ns, In (to 60m)	u	u	u	u	NCB, GCB	31
S. analis Goldenline Whiting (Shark Bay)	Oct-Apr (peak Dec-Mar)	7 months	Sand	u	Ns	u	u	u	u	NCB, GCB, WCB	34
S. ingenuua Bay Whiting	u	u	Sand	u	In (20-50m)	u	u	u	u	NCB, GCB, WCB	31
S. sihama Northern Whiting	u	u	Sand	u	Ns	u	u	u	u	NCB, GCB	31
Sparidae											
Argyrops spinifer Frypan Bream	u	u	u	u	In	u	u	u	u	GCB, NCB	
Chrysophrys auratus (Carnarvon)	Apr-Oct	7 months	Aggregations	u	Ns	u	u	u	u	GCB, WCB, SCB	36

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Snapper											
C. auratus (Metro)	Aug-Jan	6 months	Aggregations, night	u	Ns	u	u	u	u	GCB, WCB, SCB	36
Snapper											
C. auratus (Albany)	Sept-Dec	4 months	Aggregations	u	Ns	u	u	u	u	GCB, WCB, SCB	36
Snapper											
Rhabdosargus sarba	May-Dec (peak Jun-Nov)	8 months	u	u	Es, Ns	u	u	u	u	GCB, WCB, SCB	37
Tarwhine											
Scombridae											
Scomberomorus commerson	Aug-Nov	4 months	Aggregations, Reef	< 3 weeks	P, In	u	u	u	P	NCB, GCB, WCB, SCB	38, 39
Spanish Mackeral (Kimberley)											
S. commerson	Oct-Jan	4 months	Aggregations, Reef	< 3 weeks	P, In	u	u	u	P	NCB, GCB, WCB, SCB	38, 39
Spanish Mackeral (Pilbara)											
S. semifasciatus	u	u	u	u	P, In	u	u	u	P	NCB, GCB, WCB	
Grey Mackeral											

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Triakidae											
Mustelus antarcticus	Nov-Feb	4 months	u	NA	u	u	u	u	u	WCB, SCB	40
Gummy Shark											
Furgaleus macki	Mating: Aug-Sep; Ovulation: Jan-Apr	8 months	u	NA	Ns, In	Aug-Oct	3 months	u	u	GCB, WCB, SCB	41
Whiskery Shark											
Charcharhinidae											
Carcharhinus obscurus	Summer/Autumn	u	u	NA	u	Summer/Autumn	u	u	In	NCB? GCB, WCB, SCB	42
Dusky Shark											
C. plumbeus	Jan-Mar	3 months	u	NA	In	Summer/Autumn	u	u	In	NCB, GCB, WCB, SCB	43
Sandbar Shark											
C. tilstoni	Mating: Feb-Mar; Ovulation: Mar-Apr	3 months	u	NA	In	Jan	u	u	In	NCB?	44
Australian Blacktip Shark											
C. sorrah	Mating: Feb-Mar; Ovulation: Mar-Apr	3 months	u	NA	In	Jan	u	u	In	NCB, GCB	44

	Spawning					Recruitment				Bioregions	References
	Time of year	**Duration**	**Habitat**	**Pelagic larval or propagule duration**	**Aquatic zone**	**Time of year**	**Duration**	**Habitat**	**Aquatic zone**		
Spot-tail Shark											
Galeocerdo cuvier	u	u	u	NA	In	u	u	u	In	NCB, GCB, WCB, SCB	
Tiger Shark											
Negaprion acutidens	u	u	u	NA	Ns, In	u	u	u	Ns, In	NCB, GCB, WCB	
Lemon Shark											
Sphyrnidae											
Eusphyra blochii	Mating: Dec-Feb; Ovulation Mar-Apr	5 months	u	NA	u	Feb-Mar	2 months	u	u	NCB, GCB?	45
Winghead Shark											
Sphyrna zygaena	u	u	u	NA	u	u	u	u	u	NCB, GCB, WCB, SCB	
Smooth Hammerhead											
S. lewini	Mating: Sep-Dec; Ovulation Jan-Mar	7 months	u	NA	u	Oct-Jan	4 months	u	u	NCB, GCB, WCB	45
Scalloped Hammerhead											
S. mokarran	Mating: Oct-Nov; Ovulation: variable	u	u	NA	u	Dec-Jan	2 months	u	u	NCB, GCB, WCB	45
Great Hammerhead											

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Appendix 5

A summary of the traits and known effects of sediment on larval, recruiting and adult Western Australian marine fishes targeted or retained by commercial and recreational fishers. Abbreviations: u = unknown, 1 yes, 0 = no, ? = uncertain

	Effect of sediment			**Visual feeders**	**Mobile Prey**	**Clear water**	**Habitat association**	**Small home range**	**Scraping herbivores**	**Demersal eggs**	**Long PLD**	**Benthic spawning**	**Pelagic spawner**	**Benthic dwelling**
	Larvae	**Recruits**	**Adults**											
Arripidae														
Arripis georgianus Australian Herring	u	u	u	1	1	1	0	0	0	0	?	0	1	0
A.truttaceus Western Australian Salmon	u	u	u	1	1	1	0	0	0	0	?	0	1	0
Berycidae														
Centroberyx gerrardi Bight Redfish	u	u	u	1	1	1	1	?	0	0	?	0	1	1
C. australis Yelloweye Redfish	u	u	u	1	1	1	1	?	0	0	?	0	1	1
C. lineatus Swallowtail	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Carangidae														
Pseudocaranx sp. Silver Trevally	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Seriola hippos	u	u	u	1	1	1	0	0	0	0	?	0	1	? Bottom associated

Samsonfish														but move large distances
S. dumerili	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Amberjack														
Caranx ignobilis	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Giant Trevally														
Gnathanodon speciosus	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Golden Trevally														
Carangoides gymnostethus	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Bludger Trevally														
C. malabaricus	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Malabar Trevally														
Centrolophidae														
Hyperoglyphe Antarctica	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Blue-eye Trevalla														
Centropomidae														
Lates calcarifer	u	u	u	1	1	1, 0	1	?	0	0	?	0	1	1
Barramundi														
Cheilodactylidae														
Nemadactylus valenciennesi	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Blue Morwong														
Glaucosomatidae														

Glaucosoma buergeri Northern Pearl Perch	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Glaucosoma hebraicum West Australian Dhufish	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Haemulidae														
Plectorhinchus flavomaculatus Goldspotted Sweetlips	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Diagramma labiosum Painted Sweetlips	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Labridae														
Achoerodus gouldii Western Blue Groper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Bodianus frenchii Foxfish	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Choerodon rubescens Baldchin Groper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
C. schoenleinii Blackspot Tuskfish	u	u	u	1	1	1	1	?	0	0	?	0	1	1
C. cauteroma Bluespotted Tuskfish	u	u	u	1	1	1	1	?	0	0	?	0	1	1
C. cyanodus	u	u	u	1	1	1	1	?	0	0	?	0	1	1

Blue Tuskfish														
C. cephalotes	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Purple Tuskfish														
Coris auricularis	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Western King Wrasse														
Notolabrus parilus	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Brownspotted Wrasse														
Ophthalmolepis lineolatus	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Maori Wrasse														
Lethrinidae														
Gymnocranius grandoculis	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Robinson's Seabream														
Lethrinus laticaudis	u	u	u	1	1	1, 0	1	?	0	0	?	0	1	1
Grass Emperor														
L. miniatus	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Redthroat Emperor														
L. nebulosus	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Spangled Emperor														
L. olivaceus	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Longnose Emperor														
L. punctulatus	u	u	u	1	1	1, 0	1	?	0	0	?	0	1	1
Bluespotted Emperor														
Lutjanidae														

Lutjanus argentimaculatus Mangrove Jack	u	u	u	1	1	1, 0	1	?	0	0	?	0	1	1
L. carponotatus Stripey Snapper	u	u	u	1	1	1, 0	1	?	0	0	?	0	1	1
L. erythropterus Crimson Snapper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
L. johnii Golden Snapper	u	u	u	1	1	1, 0	1	?	0	0	?	0	1	1
L.lemniscatus Darktail Snapper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
L. malabaricus Saddletail Snapper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
L. russelli Moses Snapper	u	u	u	1	1	1, 0	1	?	0	0	?	0	1	1
L. sebae Red Emperor	u	u	u	1	1	1	1	?	0	0	?	0	1	1
L. vitta Brownstripe Snapper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Pristipomoides multidens Goldband Snapper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
P. typus Sharptooth Snapper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Etelis carbunculus	u	u	u	1	1	1	1	?	0	0	?	0	1	1

Ruby Snapper														
E. coruscans Flame Snapper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Symphorus nematophorus Chinamanfish	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Polynemidae														
Eleutheronema tetradactylum Blue Threadfin	u	u	u	1	1	0	0	1	0	0	0	0	1	1
Polydactylus macrochir King Threadfin	u	u	u	1	1	0	0	1	0	0	0	0	1	1
Polyprionidae														
Polyprion oxygeneois Hapuku	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Polyprion americanus Bass Groper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Platycephalidae														
Platycephalus endrachtensis Yellowtail Flathead (Swan River Estuary)	u	u	u	1	1	0	0	?	0	0	?	0	1	1
Sciaenidae														
Argyrosomus japonicas Mulloway	u	u	u	1	1	0	0	?	0	0	?	0	1	1

Protonibea diacanthus Black Jewfish	u	u	u	1	1	0	0	?	0	0	?	0	1	1
Serranidae														
Epinephelides armatus Breaksea Cod	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Othos dentex Harlequin Fish	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Epinephelidae														
Epinephelus areolatus Yellowspotted Rockcod	u	u	u	1	1	1	1	?	0	0	?	0	1	1
E. bilobatus Frostback Rockcod	u	u	u	1	1	1	1	?	0	0	?	0	1	1
E. bleekeri Duskytail Grouper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
E. coioides Goldspotted Rockcod	u	u	u	1	1	1	1	?	0	0	?	0	1	1
E. malabaricus Blackspotted Rockcod	u	u	u	1	1	1	1	?	0	0	?	0	1	1
E. morrhua Comet Grouper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
E. multinotatus Rankin Cod	u	u	u	1	1	1	1	?	0	0	?	0	1	1
E. radiates	u	u	u	1	1	1	1	?	0	0	?	0	1	1

Radiant Rockcod														
E. rivulatus Chinaman Rockcod	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Plectropomus areolatus Passionfruit Coral trout	u	u	u	1	1	1	1	?	0	0	?	0	1	1
P. leopardus Common Coral Trout	u	u	u	1	1	1	1	0	0	0	?	0	1	1
P. maculatus Barcheek Coral Trout	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Hyporthodus octofasciatus Eightbar Grouper	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Sillaginidae														
Sillaginodes punctate King George Whiting	u	u	u	?	1	1	0	?	0	0	?	0	1	1
Sillago burrus Western Trumpeter Whiting	u	u	u	?	1	1	0	?	0	0	?	0	1	1
S. vittata Western School Whiting	u	u	u	?	1	1	0	?	0	0	?	0	1	1
S. cf. robusta Stout Whiting	u	u	u	?	1	1	0	?	0	0	?	0	1	1
S. bassensis Southern School Whiting	u	u	u	?	1	1	0	?	0	0	?	0	1	1
S. schomburgkii	u	u	u	?	1	1	0	?	0	0	?	0	1	1

Yellowfin Whiting (Shark Bay)														
S. schomburgkii Yellowfin Whiting (West Coast Bioregion)	u	u	u	?	1	1	0	?	0	0	?	0	1	1
S. lutea Mud Whiting	u	u	u	?	1	1	0	?	0	0	?	0	1	1
S. analis Goldenline Whiting (Shark Bay)	u	u	u	?	1	1	0	?	0	0	?	0	1	1
S. ingenuua Bay Whiting	u	u	u	?	1	1	0	?	0	0	?	0	1	1
S. sihama Northern Whiting	u	u	u	?	1	1	0	?	0	0	?	0	1	1
Sparidae														
Argyrops spinifer Frypan Bream	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Chrysophrys auratus (Carnarvon) Snapper	u	u	u	1	1	1	1	0	0	0	?	0	1	1
C. auratus (Metro) Snapper	u	u	u	1	1	1	1	0	0	0	?	0	1	1
C. auratus (Albany) Snapper	u	u	u	1	1	1	1	0	0	0	?	0	1	1

Rhabdosargus sarba Tarwhine	u	u	u	1	1	1	1	?	0	0	?	0	1	1
Scombridae														
Scomberomorus commerson Spanish Mackeral (Kimberley)	u	u	u	1	1	1	0	0	0	0	?	0	1	0
S. commerson Spanish Mackeral (Pilbara)	u	u	u	1	1	1	0	0	0	0	?	0	1	0
S. semifasciatus Grey Mackeral	u	u	u	1	1	1	0	0	0	0	?	0	1	0
Triakidae														
Mustelus antarcticus Gummy Shark	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
Furgaleus macki Whiskery Shark	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
Charcharhinidae														
Carcharhinus obscurus Dusky Shark	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
C. plumbeus Sandbar Shark	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
C. tilstoni Australian Blacktip Shark	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
C. sorrah	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0

Spot-tail Shark														
Galeocerdo cuvier Tiger Shark	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
Negaprion acutidens Lemon Shark	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
Sphyrnidae														
Eusphyra blochii Winghead Shark	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
Sphyrna zygaena Smooth Hammerhead	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
S. lewini Scalloped Hammerhead	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0
S. mokarran Great Hammerhead	u	u	u	1	1	1	0	0	0	0	NA	NA	NA	0