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A critical analysis of the direct effects of dredging on fish

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Abstract

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

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

1 | INTRODUCTION

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

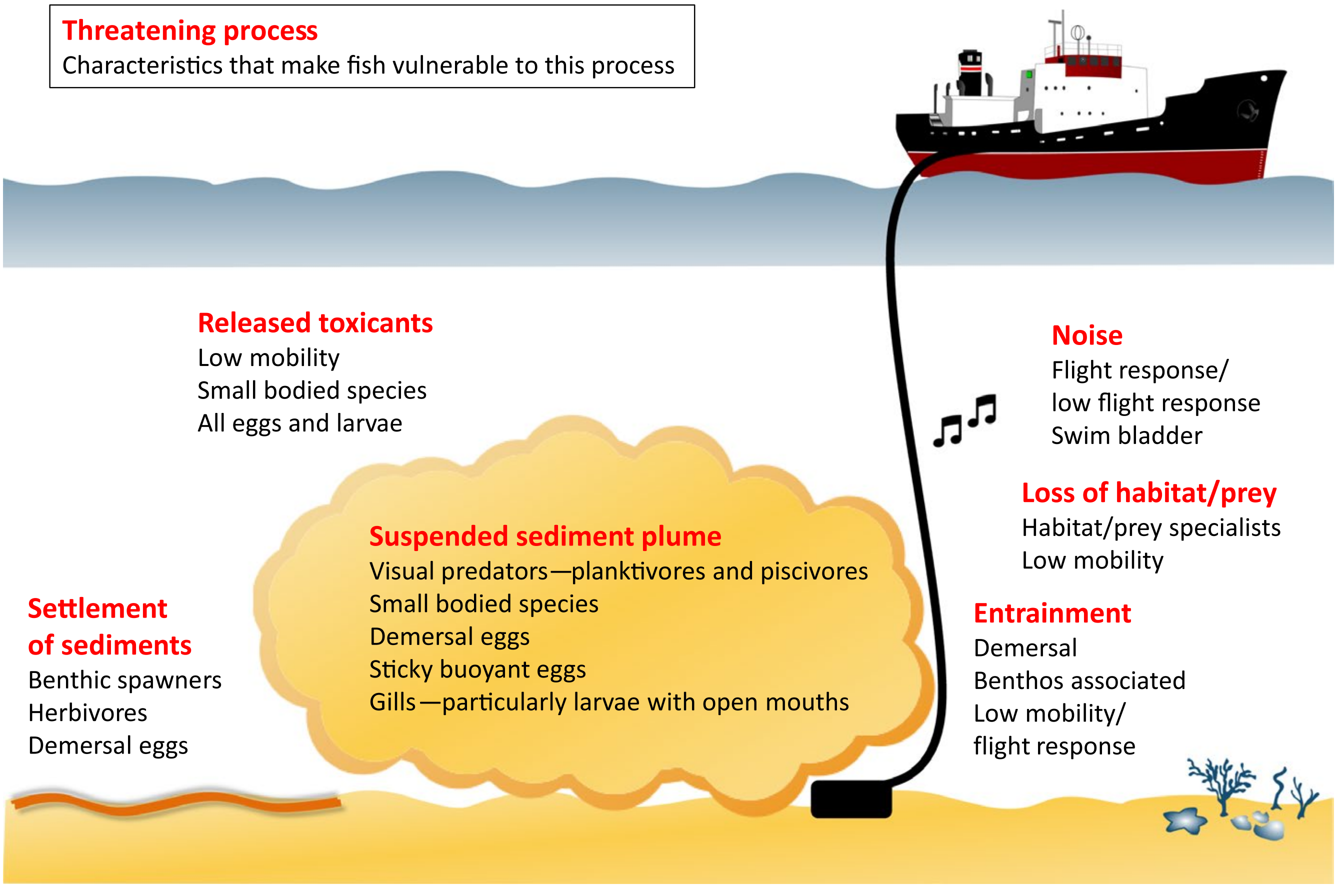

FIGURE 1 A schematic diagram of categories of potential effects of dredging on fish. [Colour figure can be viewed at wileyonlinelibrary.com]

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).

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

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,

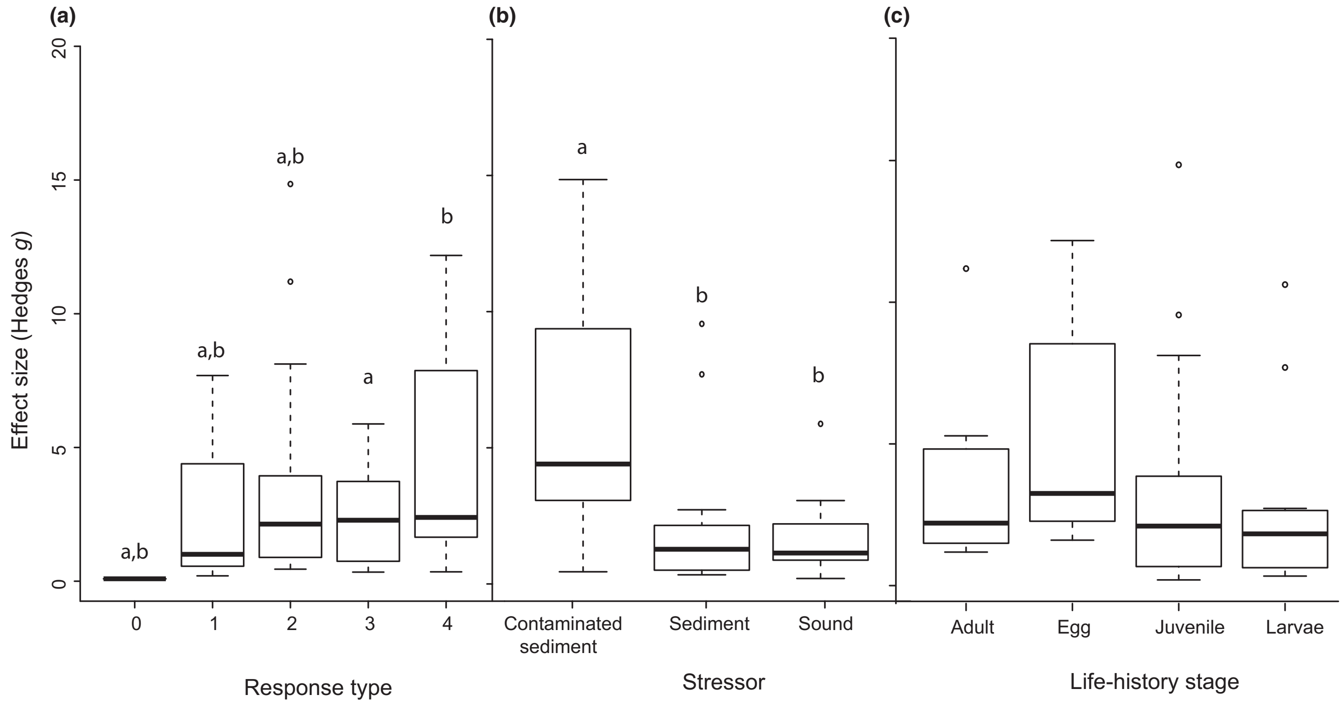

FIGURE 2 The impact of (a) response type, (b) stressor type, (c) life-history stage on effect size across all stressors. A response type of 0 = no effect, 1 = minor behavioural changes, 2 = minor physical damage or moderate behavioural changes, 3 = physiological impacts and 4 = increased mortality. Variables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

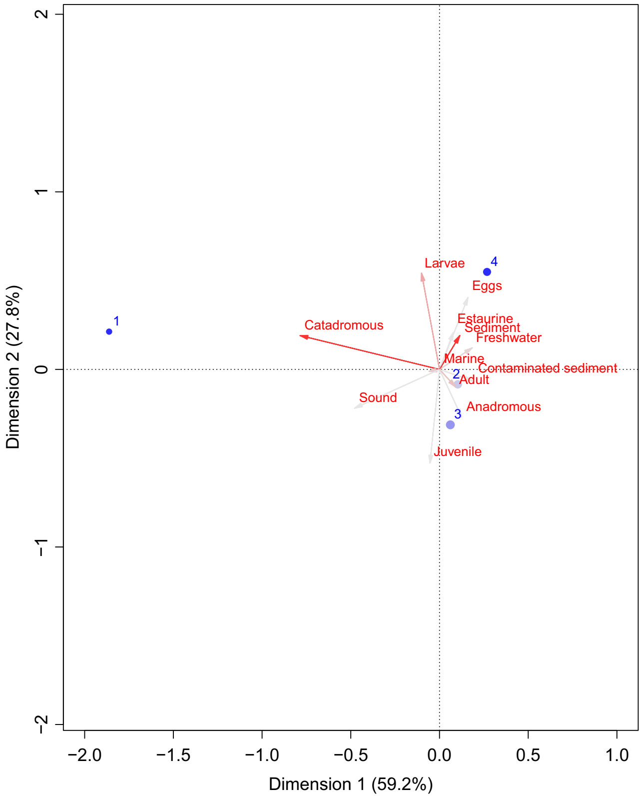

FIGURE 3 An asymmetric graph of the linear correspondence analysis, with the response type in the principal coordinates and the explanatory variables in reconstructions of the standardized residuals (square root of the relative frequency). Response type is represented by points, and the explanatory variables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

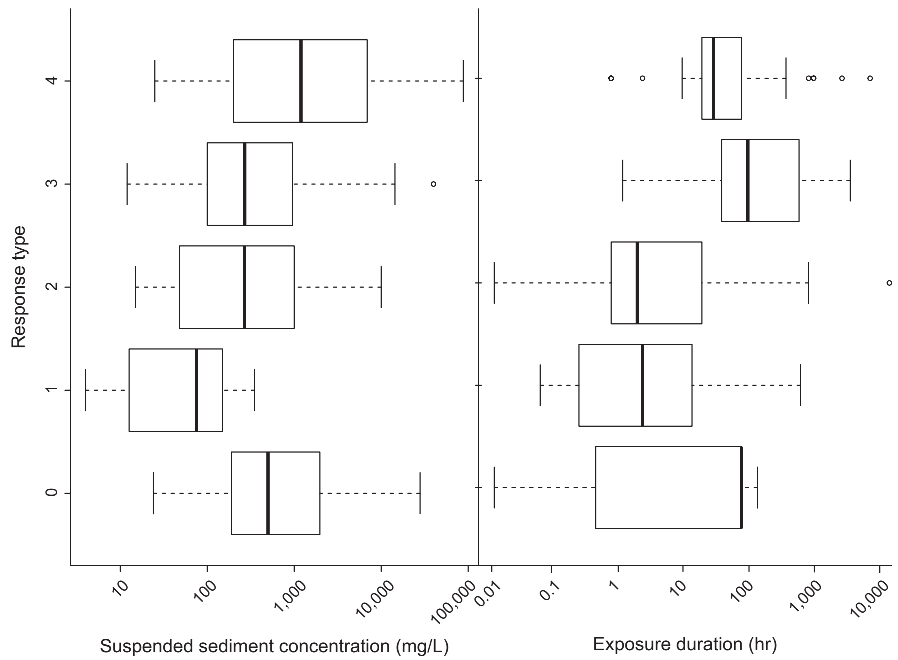

FIGURE 4 The impact of (a) suspended sediment concentration and (b) exposure duration on the type of effect elicited by suspended sediment. A response type of 0 = no effect, 1 = minor behavioural changes, 2 = minor physical damage or moderate behavioural changes, 3 = physiological impacts and 4 = increased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