GOVERNMENT OF
WESTERN AUSTRALIA

# Technical Guidance

## Environmental impact assessment of marine dredging proposals

Environmental Protection Authority

September 2021

| Version | Change | Date |
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| 1.0 | Initial version | December 2016 |
| 2.0 | Technical guidance includes the findings of the WAMSI Dredging Science Node relating to tolerance thresholds of key benthic marine organisms to dredging pressures and on critical life cycle windows where important marine taxa are likely to be particularly sensitive to these pressures. The structure and content of the main document is largely retained, with the detailed scientific information included in three technical appendices. | September 2021 |

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Cover image: Dredge off Fremantle © Ray Masini 2010

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# 1 Introduction and background

## 1.1 Purpose

This technical guidance describes an impact prediction and assessment framework that the EPA encourages proponents to use so that predictions of impacts to benthic habitats associated with dredging activities referred to the EPA are presented in a clear and consistent manner. This guidance is focussed on describing the effects on benthic habitats caused by removal/burial at the sites of dredging and disposal, and the effects of suspended and deposited sediments further afield. These impacts should then be considered in the context of the EPA's guideline and technical guidance for the environmental factor 'benthic communities and habitats' (EPA 2016d) to ensure consistency with the factor objective. Proponents should also ensure that these issues are addressed in the broader context of the overarching environmental protection principles set out in section 4 of the *Environmental Protection Act 1986*. This guidance doesn't address the potential effects of turbidity on recreational uses such as swimming, or disturbance and release of toxicants and other contaminants associated with dredged sediments that may affect seafood quality[1], or other environmental issues such as potential effects on coastal processes or marine fauna[2] that may be associated with dredging activities. Furthermore, this guidance does not address issues associated with seeking approvals under the Commonwealth *Environmental Protection (Sea Dumping) Act 1981*.

The framework applies across Western Australia (WA) and does not differentiate between types of dredging proposals or consider regional environmental differences.

Section 3 of this guidance sets out a methodology that proponents can use for impact prediction, assessment and management of dredging proposals on marine biota – particularly benthic communities. Section 3.4 sets out the zonation scheme approach that the EPA expects proponents to use to present their impact predictions. The general approach is not new and has been recommended for use by the EPA and applied since 2011 (EPA 2011, 2016c). It has been found to be robust and to provide clarity and consistency in the environmental impact assessment (EIA) of dredging proposals in the face of high levels of uncertainty.

The recent completion of the research program undertaken by the Dredging Science Node of the Western Australian Marine Science Institution (WAMSI DSN) (https://www.wamsi.org.au/dredging-science-node) has significantly increased the level of understanding of dredging pressures, and the tolerance of marine biota to those pressures, in the tropical waters of north-west Western Australia (NW WA). The results have been made available through the publication of nearly 100 scientific reports including 53 peer-reviewed journal articles[3].

This is an enormous amount of information, and to assist proponents to interpret and apply it efficiently and consistently, the key relevant findings have been identified and further refined with input from regulators and experienced environmental consultants. The intent of this work was to develop a set of scientifically-based guidelines and approaches that clearly align with the EPA's recommended approach for presenting impact predictions, and assist proponents in the EIA and management of dredging programs in WA.

The outcomes of this process have been outlined in sections 3.5–3.7 of this guidance and more detail, including suggested guideline values and their rationales, is provided in the appendices. It is important to note that the guidelines in the appendices are not prescriptive, rather they can be considered a set of default guidelines that can be used in the absence of more robust site-specific information.

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1 See EPA (2016b) *Technical Guidance – Protecting the quality of Western Australia's marine environment* for technical advice on the potential effects of toxicants and other contaminants.

2 See EPA Environmental Factor Guidelines on coastal processes and marine fauna for more information and advice.

3 The publications are available at https://www.wamsi.org.au/dredging-science-node

Proponents are ultimately responsible for the approaches they adopt for impact prediction and management, and ensuring they are robust and appropriate for the environmental setting and the nature and scale of their dredging projects.

Proponents are strongly encouraged to seek early advice from suitably qualified specialists and the EPA Services Directorate of the Department of Water and Environmental Regulation regarding the application of this guidance, including the use of predictive numerical simulation models, in the context of their proposals.

It should be noted that while the framework outlined in this guidance is focussed on the EIA of dredging-related activities, the approach can also be adapted and applied to the EIA of other types of development proposals where there is significant uncertainty around impact predictions (see sections 3.4 and 3.6).

## 1.2 Background

Dredging is an activity carried out to ensure safe vessel access at existing and new ports, harbours and other coastal facilities. Sediment plumes associated with dredging have the potential to influence large areas beyond the direct footprint of the development. The scale and number of significant dredging projects in WA has been large by world standards, and this places the EPA at the forefront of dealing with the environmental issues associated with these types of development. In addition to the large scale of dredging and the potential spatial extent of its influence, dredging projects often occur in sensitive environments with unique and/or generally poorly-understood biodiversity and ecology (e.g. understanding of the natural tolerances and susceptibilities of key biota). This uncertainty presents significant challenges for environmental impact assessment and management.

To reduce the uncertainty with dredging EIA and management, the Government of Western Australia required the proponents of three large capital dredging projects to provide funding for research to improve understanding of dredge related pressures and of the response of marine biota to those pressures. These funds were subsequently combined to establish the WAMSI DSN, and the $9.5 million provided by proponents was used to leverage a further $9.5 million into the project from research providers. Valuable environmental monitoring datasets from a number of large-scale dredging programs in NW WA were also made available by proponents which greatly assisted the research effort. The findings provide a strong evidence base for the prediction and management of dredging impacts by proponents and regulators. These findings have been further synthesised into relevant guidelines for the EIA of dredging proposals on benthic communities and habitats and incorporated into this version of the EPA's *Technical Guidance – Environmental impact assessment of marine dredging proposals*.

The framework adopted here by the EPA takes a pragmatic approach that ensures the range of likely impacts are considered in EIA based on sound scientific principles. The adoption of key findings from the WAMSI DSN should lead to more robust and precise impact assessments, and reduce the post-assessment monitoring and management burden on proponents because of the increased confidence in resultant impact predictions and management. Furthermore, this framework encourages proponents to incorporate any new knowledge of pressures, and impacts and responses to those pressures, to refine and improve their predictive models.

Other legislation, regulations, management frameworks and guidance also exist for a number of key environmental issues relevant to the assessment, management and regulation of dredging proposals. These environmental issues include sea dumping, contaminated site assessments and protection of wildlife. It is the responsibility of proponents to address the requirements of all relevant legislative and regulatory frameworks and guidance issued by other agencies. The EPA draws upon information presented by proponents in the context of these (and other) relevant regulatory frameworks and the advice of relevant regulators during its assessment of dredging proposals.

# 2 Context

## 2.1 What is dredging?

Dredging involves excavation of the seabed, typically underwater, but may also occur in intertidal areas during low tide or behind constructed bunds designed to maintain a 'dry' dredge site.

A number of different types of dredges are typically used for dredging proposals in WA. These include hydraulic dredges such as cutter suction dredges and trailing suction hopper dredges, and mechanical dredges including bucket or grab dredges.

Most dredging proposals are carried out to provide navigable water depths for shipping in ports and harbours and associated shipping channels. Dredging of trenches for the placement of subsea pipelines, and subsequent backfilling, is another relatively common practice. Dredging for marine mining operations that target calcium carbonate, diamonds and other resources is also proposed from time to time.

> For the purpose of this document, dredging refers to seabed excavation and dredge material placement activities that introduce sediments to the water column.

Once material is excavated from the seabed by a dredge, it can be handled in a number of different ways. Often dredged material is loaded into a hopper (part of the dredge itself or on a separate vessel) and transported to a disposal site where the contents of the hopper are emptied directly in the open ocean (i.e. *sea dumping*) or via a pipeline that allows the dredge material to be pumped to a location where it is used for 'alternative' purposes (e.g. *land reclamation*). Depending on the type of equipment being used and the substrates involved, dredged material is sometimes pumped directly from the dredge site to a disposal location on land or at sea.

Material dredged for pipeline trenches is sometimes placed temporarily on the seabed adjacent to the trench (i.e. side-cast) before being placed back into the trench to stabilise and protect the pipe after it has been laid. Less commonly, some dredging operations for port facilities involve dredged material being side-cast near the dredge site before it is picked up by another dredge and transported to the disposal site.

## 2.2 Environmental considerations

All dredging causes an environmental impact at dredge and disposal sites (Victoria EPA 2001, EPA 2013, Mills and Kemps 2016, WAMSI 2019) and potentially further afield (PIANC 2010). Some examples of the types of potential impacts associated with dredging proposals include:

*Impacts to benthic communities and habitats addressed by this guidance*

- direct loss of benthic communities and habitats by removal or burial
- indirect impacts on benthic communities and habitats from the effects of sediments introduced to the water column by the dredging and dredge spoil disposal.

*Other types of impacts that are not part of this guidance*

- effects of suspended sediment and increased turbidity on fish behaviour, visual acuity, gill function and survival (this issue was considered in Theme 8 of the DSN)
- changes to shorelines, bathymetry and habitats through modified ecological and physical processes (this issue is considered in the *Environmental Factor Guideline – Coastal processses* (EPA 2016e))

- introduction of invasive pest species translocated in dredging (or ancillary) equipment that can have both ecological and economic consequences (responsibility for addressing this issue rests with the Department of Primary Industries and Regional Development)
- adverse effects of contaminant release from sediments and dispersion (including impacts associated with reclamation or onshore disposal of acid sulphate soils) on marine environmental quality (this issue is addressed through the *Technical Guidance – Protecting the quality of Western Australia's marine environment* (EPA 2016b))
- changes to coastal processes and water circulation that impact on the environmental values of the coast and coastal waters (this issue is considered in the *Environmental Factor Guideline – Coastal processes* (EPA 2016e))
- impacts on the behaviour and survival of marine wildlife, including specially protected species (this issue is considered in the *Environmental Factor Guideline – Marine fauna* (EPA 2016f)) .

> This document only provides guidance for the presentation of predicted impacts of dredging activities on benthic communities and habitats caused by direct removal, burial or indirect impacts of suspended sediments (excluding chemical contaminants).

Although the other types of impacts listed above are not addressed further in this document, this should not be taken to imply that they are not relevant or important. Proponents should refer to the EPA's guidelines and technical guidances for other environmental factors relevant to the marine environment when considering these types of impacts. In some locations, dredging may have implications for marine conservation reserves and/or marine fauna, or for public uses of the environment such as commercial and/or recreational fishing and tourism. Where dredging involves contaminated sediments the disposal of those sediments could create a contaminated site which may need to be regulated[4].

### 2.2.1 Dredge-generated sediments and their effects

Dredging and spoil disposal introduces sediment to the water column to varying degrees from three main sources:

1. mechanical interaction of the dredging equipment with the seabed substrates
2. overflow associated with loading[5] of dredged material and land reclamation
3. disposal of dredge spoil at sea.

The mechanical interaction of dredging equipment with the seabed causes sediment particles, in a range of particle sizes, to be introduced to the surrounding water column at the dredge site (e.g. loss from the cutting head of a cutter suction dredge or spillage from grab/bucket dredges). Limited under-keel clearance and turbulence from propellers can also disturb and lift sediments into the water column.

Hydraulic dredges produce slurries that comprise a fine sediment-water mixture and dredged solids. When the fine sediment-water mixture is allowed to escape during loading at the dredging site or from a land reclamation area, it can introduce significant loads of fine sediment to the water column. This sediment-laden discharge is the second principal source of sediment introduced to the water column by dredging and is commonly referred to as *overflow* or *spill* when discharged from vessels or *return water* when discharged from reclamation areas (see Mills and Kemps 2016 for overview).

[4] In this circumstance advice should be sought from Contaminated Sites Branch in DWER.

[5] As defined in the *National Assessment Guidelines for Dredging*, Commonwealth of Australia, Canberra, 2009.

Some sediment is also introduced to the water column during disposal of dredged material at sea, although the proportion of fines retained in spoil is relatively low when overflow practices are used during loading. Accordingly, in many cases only a relatively modest proportion of all fine sediments produced by dredging is introduced to the water column during dumping at sea. Exceptions to this will arise where overflow at the dredge site is eliminated or highly controlled to manage release of contaminants or when dredging up-current of particularly important areas.

The effects of dredge generated suspended sediments on benthic communities and habitats are considered to be indirect effects in the context of this guidance. The primary indirect environmental effects (see section 3.2) relate to:

1. decreased light transmission through the water column reducing the amount of light available at the seabed, leading to a lowering of primary production and even death of benthic primary producers if effects are acute or prolonged
2. increased rates of sediment deposition beyond natural levels leading to stress and in extreme cases mortality, and to a lesser extent
3. abrasion of membranes or clogging of breathing or filter feeding organs on some benthic invertebrates causing stress and even death of more sensitive species.

The characteristics of sediment introduced to the water column by dredging can be very different to the characteristics of natural substrates and suspended sediments at a dredge site. The characteristics of sediments generated and released by dredging is influenced by a range of factors including the geotechnical characteristics of the substrates to be dredged, the type of dredge and its mode of operation, and the nature of the interaction between the dredge and seabed substrate.

Far-field dredge plume modelling is generally used to assess the potential extent, duration and degree of influence of the plumes on water quality and biological communities. Dredge source-terms are used to estimate the suspended sediment loads entering the far-field from dredging activities, but there is significant uncertainty and variation surrounding these terms and little quantitative study of sediment release rates from dredging activities into the far-field. Kemps and Masini (2017) and Sun et al (2019) provide reviews and recommended approaches for far-field source term estimation.

> Predicting impacts of dredge-generated sediments relies on understanding the key factors that influence the generation, sources, physical characteristics and release rates of fine sediments.

# 3 Methodology

## 3.1 General approach

In the first instance the proponent assessment documentation should detail how the steps to impact mitigation described below have been considered in advance of presenting predictions of environmental impact.

1. There should be demonstrable consideration of options to avoid impacts on benthic communities due to dredging, for example, by providing the rationale for selection of the preferred site and the proposed dredging methods and their timing (see section 3.7).
2. Where impacts cannot be avoided, then proposed project design should aim to minimise impacts (e.g. through iterative design and demonstrable application of principle 3 below) and the proposed design should be justified in terms of operational needs and environmental constraints of the site.
3. Best efforts should be made to demonstrate in EIA documentation that all 'reasonable and practicable measures'[6] have been taken to prevent or minimise impact, including through design, selection of construction methods and environmental management aimed at minimising predictive uncertainty and environmental impacts.

The level to which proponents demonstrate how they have considered impact avoidance and minimisation (consistent with the mitigation hierarchy outlined in the EPA's EIA rocedures Manual) and application of all reasonable and practicable measures to prevent or minimise impacts in all aspects of their proposals, will be considered when assessing whether the proposal is consistent with EPA objectives.

The assessment framework described in this guidance is designed to impart clarity and consistency to the way predicted impacts are presented to the EPA for assessment. It establishes an approach for generating and presenting predictions of *the likely range* of environmental impacts, which in turn, provides the basis for facilitating the transfer of these predictions into recommended conditions and environmental monitoring and management strategies.

In simple terms, the predictions are made by superimposing the dredging pressures (i.e. excavation, burial, sediment deposition, elevated suspended sediment concentrations/turbidity and reduced light availability) on the biological communities and determining the likely responses of communities to those pressures.

While it is not the intention of the EPA to mandate a specific methodology, in order to generate realistic impact predictions, proponents are encouraged to consider and apply guidance provided in the following sections:

- describing benthic habitats (section 3.1.1)
- background environmental data (section 3.1.2)
- describing impacts (section 3.2)
- generating and representing predictions (sections 3.3–3.5)
- integrating predictions with monitoring and management (section 3.6).

[6] Some examples of 'reasonable and practicable measures' are outlined in section 3.7.

More detailed guidance is provided in Appendices A and C for impact prediction and management respectively. The known information on the timing of reproduction/recruitment of a range of WA marine taxa is presented in Appendix B. The focus is on corals but also includes information on fish, invertebrates, seagrasses and macroalgae. However, as described in section 3.7, critical windows of sensitivity apply to a wide range of species including listed and protected species. Proponents are encouraged to consider this information when designing dredging programs to reduce the risk of adverse effects on these key life-cycle processes where practicable.

### 3.1.1 Describing benthic habitats

An adequately detailed benthic habitat map is a critical piece of information for assessing the impacts associated with dredging.

The benthic habitat map (or series of maps) supplied by proponents must be reasonably up to date and be at a sufficiently fine scale to provide confidence in the habitat boundaries which in turn reduces uncertainty in relation to the predictions of the areas of impact. Mapping should be undertaken as finely and accurately as possible considering the primary purpose and end use of the maps (e.g. to evaluate habitat loss and inform location of monitoring and reference sites). Factors such as expected intensity of pressure and the types and uniformity (or heterogeneity) of existing biological communities should also be considered. For example, the main benthic habitat types might be defined on the basis of the abundance of dominant and sub-dominant functional groups.

Spatial coverage of benthic habitat surveys and mapping is an important consideration. As a general rule, mapping coverage should extend across any predicted Zones of High and Moderate Impact and the area of the Zone of Influence[7] immediately outside of the Zone of Moderate Impact. Some level of mapping may also be required across any local assessment units established to assess cumulative impacts to benthic communities and habitats (EPA 2016a). High quality data on the extent and distribution of benthic habitats in the Zone of Moderate Impact and adjacent Zone of Influence will be necessary for identifying suitable monitoring sites to manage environmental performance and assess compliance during project implementation. Knowledge developed through the survey work will also inform the selection of local biota that may be suitable surrogates or indicators for impact prediction and monitoring. Appendix C provides more detailed advice on mapping coral, seagrass and filter-feeder communities.

Technical reports that describe how benthic habitat surveys and mapping were conducted and how maps were produced must be supplied as part of the EIA documentation. Reports should clearly state any assumptions and consider their implications, and describe methodologies including those employed in the field for surveys and in the office to interpret data and prepare spatial products. Spatial data associated with the benthic habitat map(s) and infrastructure outlines should be supplied to the EPA in a suitable GIS compatible format.

> Proponents are required to submit IMSA[8] data packages to the EPA that accompany marine benthic habitat survey reports. Instructions and templates are available on the EPA website to assist proponents in this regard.

Historical data can provide useful information to design cost-effective surveys and a basis to evaluate the relative stability of those habitats and dominant biota over time. Historical and more contemporary data collected by other proponents may be available for some or all of the area of the proposal and proponents are encouraged to utilise the available information to consolidate and improve knowledge of these habitats. This may also substantially reduce the extent of, or in some cases eliminate the need

[7] The terms Zone of High Impact, Zone of Moderate Impact and Zone of Influence are described in section 3.4.

[8] The Index of Marine Surveys for Assessment (IMSA) is an online portal for the systematic capture and sharing of marine data created as part of an environmental impact assessment.

for, detailed project specific habitat mapping. Proponents are encouraged to interrogate IMSA for benthic habitat data relevant to their proposal area.

An understanding of the current and historical extents and distribution of benthic habitats is an integral requirement for the EIA of marine dredging proposals. Descriptions and maps of the different benthic habitats should be fit-for-purpose and accompanied by clear descriptions of the methods used to generate them.

### 3.1.2 Background environmental data

Acquisition and analysis of background data is an integral part of any environmental impact assessment. For example, long-term background data sets for a suite of dredging-relevant environmental variables (e.g. underwater light climate, suspended solids concentration (SSC), sediment deposition rate, correlations between these factors) can be used to develop knowledge about natural tolerances and susceptibilities of local benthic organisms. Furthermore, baseline data sets are critically important for calibration and validation of numerical models (see Sun et al 2019).

Historical local or regional data can provide useful information to support the impact assessment and, if contemporary data are required, support the design of cost-effective data collection programs. Historical data in some instances may substantially reduce the extent of, or in some cases eliminate the need for, detailed project specific data collection. In addition, advances in remote sensing analysis techniques should be considered to assess historical water quality datasets from archived satellite imagery (see Fearns et al 2019). Proponents are encouraged to interrogate IMSA for background environmental data relevant to their proposal area.

The types of background data required, and how they should be collected and presented, will be strongly influenced by the environmental setting of the proposal. The dominant pressure-effect pathways that link sediments suspended by dredging to the local biota are of particular importance. The WAMSI DSN found that light availability at the seabed is a critical indicator of dredging-related pressure followed by sediment deposition. Both of these variables were associated with SSC (often measured as turbidity). Inshore waters are typically more turbid than offshore waters, particularly near river mouths and wide intertidal mudflats. Natural selection will favour species (and potentially genotypes) that are tolerant of the ambient conditions and select against those that are not.

It is possible that guidelines derived for offshore, clear water communities are naturally exceeded in near-shore turbid waters and hence if applied would likely be overly protective and lead to overestimation of impacts. Conversely, applying guidelines derived for turbid water species in clear water situations would be likely to underestimate impacts. In both these situations alternative site-specific guidelines could be derived from baseline data collected for the critical indicator(s) at the site (e.g. light) and using the recommended approaches in the *Technical Guidance – Protecting the quality of Western Australia's marine environment* (EPA 2016b). There may also be a need to collect baseline biological health/condition data prior to construction to establish natural levels of stress response and enable post-construction impact assessments where appropriate. Appendix C provides more detailed advice on establishing environmental baselines for coral, seagrass and filter-feeder communities.

Proponents are strongly encouraged to seek specialist professional advice regarding the types of baseline data that should be collected to inform the selection of relevant and appropriate guideline values and improve confidence in any predictions of the extent, severity and duration of dredge-related environmental impacts.

Relevant background environmental data should be used to inform, validate and enhance confidence in predictions of environmental impacts.

## 3.1 Describing impacts

EIA is based on predictions of the extent, severity and duration of environmental impacts, taking into account confidence that can be placed in the predictions and the likely effectiveness of proposed monitoring and management strategies.

The EPA expects that both direct and indirect impacts are considered explicitly.

Direct impacts occur predominantly within and immediately adjacent to infrastructure footprints where dredges excavate the seabed and where rock armour and spoil is dumped. Direct impacts typically involve irreversible loss or serious damage to benthic habitats and communities, where *serious damage* means 'damage to benthic communities and/or their habitats that is effectively irreversible or where any recovery, if possible, would be unlikely to occur for at least 5 years' (EPA 2016a).

Indirect impacts can include serious damage, reversible impacts and physiological effects. They are generally caused by dredge-generated suspended sediment plumes that extend over areas surrounding infrastructure footprints, dredging sites and spoil disposal sites and occur when sediment deposition rates and/or elevated turbidity exceed the natural tolerance levels of benthic communities exposed to those pressures. The findings of the WAMSI DSN[9] suggest that the key dredging-related pressures on biota are reduction in available light and sediment deposition, both of which are a function of the amount and types of sediment that become suspended in the water column. These indirect effects of dredge-generated suspended sediments should be the focus of impact prediction, but they are perhaps the most challenging to predict given that background conditions are in constant flux due to natural processes. Appendix A provides more detailed discussion of these issues and guidelines to assist in predicting the indirect effects of dredge-generated sediments on key benthic communities.

Impact predictions cannot meaningfully consider the effects of dredging in isolation of natural background conditions.

> The relevant pressure experienced by the biota is the cumulative total arising from the simultaneous effects of natural processes and dredging-induced changes and so predicted impacts should be based on cumulative pressure.

Recent research and in-depth analyses of dredge impact monitoring of large-scale dredging programs in the Pilbara have shown the impacts typically come about due to a combination of smothering and turbidity-related effects in the near field close to dredging and disposal areas (10s to 100s of metres), and from turbidity-related effects in the far field that may extend over much larger distances (kilometres) down-current from the suspended sediment generating activities. The health of the biota may be affected directly (e.g. through chronic light depravation at the seabed caused by excess turbidity) or through effects on key ecological processes they rely on for survival at the community level such as reproduction and/or recruitment.

WAMSI DSN research found that some functional groups are more susceptible to a given frequency, intensity and duration of pressure than others. Corals and seagrasses for instance are generally more susceptible to light depravation than sponges. Even within functional groups there are differences in susceptibility based on morphology. For example, branching corals were found to be less susceptible to sedimentation than massive or encrusting species, but more susceptible to light depravation. Furthermore, some genera within functional groups are more or less susceptible than others. An example is the coral genus *Porites* that has a physiological mechanism to dislodge deposited sediments enabling it to survive sediment deposition events that other coral genera could not. On the other hand, *Porites* are very slow growing and, if lost, these communities may take decades or longer to recover.

9 See WAMSI (2019) for overview.

Conversely, *Acropora* sp. expend proportionally more energy on reproduction and to achieve fast growth which make them more susceptible to prolonged adverse conditions. However, these same attributes enable comparatively rapid recovery from residual fragments or recruitment once conditions become favourable again.

On average the pressures and resultant effects would typically attenuate with distance from the source, consequently impacts on benthic species and communities range in severity and duration from serious and irreversible damage to barely measurable and readily-reversible effects (see Figure 2).

> Both direct and indirect impacts, along with an assessment of the seriousness/reversibility of those impacts, are to be included in predictions of impacts associated with dredging proposals.

## 3.3 Generating predictions

### 3.3.1 General

Predicting direct impacts of dredging is relatively straightforward as these impacts are generally tightly linked to the dredge area and/or disposal sites and immediate surroundings.

Numerical modelling is most commonly used to predict the extent, intensity and persistence of dredge-generated sediment plumes, and the extent, severity and duration of resultant indirect impacts in benthic habitats. The EPA recognises the modelling of dredging-related pressures and biological effects is challenging, but that it provides important and useful information on the likely nature of sediment plumes generated by the proposal and their likely environmental impacts. Therefore, numerical modelling will continue to be an integral component of the EIA of dredging proposals (PIANC 2010, DEMG 2011, GBRMPA 2012).

In very simple terms the approach commonly applied to predict indirect impacts from dredge-generated sediments involves implementing three key types of predictive modelling in a logical sequence:

- hydrodynamic modelling
- sediment transport modelling
- ecological response modelling.

Proponents should consider relevant contemporary approaches to predictive modelling. For example, the WAMSI DSN guideline on dredge plume modelling for environmental impact assessment (Sun et al 2019) provides guidance and recommendations relevant to the modelling of dredge plumes in the Western Australian context. It discusses how uncertainty can be addressed and provides advice on how to calibrate, validate and parameterise numerical hydrodynamic and sediment transport and fate models, and account for natural and dredge-generated sediment transport pathways. It highlights the importance of identifying key pressure-response pathways that mediate impacts of dredging on key benthic biota, to inform the physical modelling strategy and ensure the outputs are fit-for-purpose.

> Direct impacts are generally predicted based on a combination of information about the areas to be dredged and disposal areas. The extent, severity and duration of indirect impacts are generally predicted with the use of simulation models, sometimes supplemented with empirical data collected during previous dredging projects. Proponents should seek early advice from suitably qualified specialists and the EPA Services Directorate on the use of predictive numerical simulation models in the context of their proposals.

### 3.3.2 EIA and modelling

Clearly presented information regarding calibration and validation of numerical models, assumptions and sources of uncertainty, and their associated implications for predictions, will assist the EPA in forming judgements about the reasonableness of the predicted environmental impacts, and the confidence it can place on those predictions.

The level of agreement between model outputs and data measured in the field will vary from application to application and depend on many factors. It is also important to note that the biological effects guideline values for suspended sediment concentration (and related parameters) presented in this guidance are for absolute levels of suspended sediment (i.e. natural + dredging-related), not just from dredging-related activities. As such modelling of ambient sediment dynamics is likely to be required to generate ecologically-relevant pressure fields for interrogation against the guidelines. Modelling (and validation) of ambient sediment dynamics can be done in the absence of detailed dredging plans; modelling of absolute levels relies on detailed dredging plans, and validation can only occur after dredging commences. For these reasons the EPA has not specified the level of agreement between model outputs and observations to be achieved. Instead, the EPA expects proponents to set out the process and outcomes of calibration and validation exercises and relevant assumptions on a project-by-project basis.

Proponents are encouraged to consider the recommendations set out in Sun et al (2019) for issues surrounding model set up and calibration, including the collection of appropriate field data and incorporating natural sediment dynamics. A key parameter that is particularly difficult to establish relates to the 'source term' that in simple terms relates to the amount of suspended sediment input to the model that is used to represent dredging-related sediment generation. It will be highly dependent on the substrates being dredged and on the types and modes of operation of the dredging equipment. Sun et al (2019) provides advice on how to first estimate source terms (e.g. for EIA purposes) and then obtain direct measurements to refine management strategies.

There are very few reliable and publicly available data derived from direct measurements of actual suspended sediment characteristics and concentrations generated by dredging. The development of a publicly accessible data base of source terms for various dredger-substrate combinations developed through direct measurement during dredging campaigns in Western Australian waters would greatly improve confidence in this element of the simulation modelling program to support EIA (see Kemps and Masini 2017 and Sun et al 2019).

> To improve confidence in dredging EIA, numerical models should be calibrated and validated and any associated assumptions and implications of those assumptions should be clearly stated and evaluated.

In cases where all relevant proponent documentation is not provided, is ambiguous or includes unsubstantiated conclusions, the level of confidence in the prediction would generally be lower than if high quality, peer reviewed information is provided.

### 3.3.3 Peer review

While the EPA does not routinely require proponents to commission peer reviews of studies underpinning EIA, in some situations peer reviews by suitably qualified experts may assist the EPA in achieving timely assessments. If proponents either choose to commission a peer review or are requested to do so by the EPA, it is beneficial to seek advice and agreement with the EPA Services Directorate on the terms of reference and scope before commencing the review.

To maximize the effectiveness and transparency of the peer review process, the EPA expects to receive the peer reviewer's reports, including their 'close out' comments based on the document that is ultimately submitted for EIA.

Proponents should note that information relating to the peer review, including the terms of reference and the peer reviewer's reports, may be made publicly available as part of the EIA process.

## 3.4 Describing impact predictions

### 3.4.1 Impact zonation scheme

The EPA has developed a spatially-based zonation scheme for proponents to use as a common basis to describe the predicted extent, severity and duration of impacts associated with their dredging proposals. The scheme consists of three zones that represent different levels of impact:

- **Zone of High Impact (ZoHI)** is the area where serious damage to benthic communities is predicted or where impacts are considered to be irreversible. The term *serious damage* means 'damage to benthic communities and/or their habitats that is effectively irreversible or where any recovery, if possible, would be unlikely to occur for at least 5 years'. Areas within and immediately adjacent to proposed dredge and disposal sites are typically ZOHI. The loss of the benthic communities and/or habitats within these zones should be considered irreversible, unless a defensible case for recovery of the impacted benthic communities and habitats can be presented.
- **Zone of Moderate Impact (ZoMI)** is the area within which predicted impacts on benthic organisms are sub-lethal, and/or the impacts are recoverable within a period of 5 years following completion of the dredging activities. This zone abuts, and lies immediately outside of, the ZOHI. Proponents should clearly explain what would be protected and what would be impacted within this zone, and present an appraisal of the potential implications for ecological integrity of the impacts over the timeframe from impact to recovery (e.g. through loss of productivity, food resources, shelter). Where recovery from the impact predicted in this zone is likely to result in an 'alternate state' compared with that present prior to development, then this outcome should be clearly stated in environmental assessment documents, along with justification as to why the predicted impacts should be included within this zone (rather than the ZOHI) and an appraisal of the potential consequences for ecological integrity. The outer boundary of this zone is coincident with the inner boundary of the next zone, the Zone of Influence.
- **Zone of Influence (ZoI)** is the area within which changes in environmental quality associated with dredge plumes are predicted and anticipated during the dredging operations, but where these changes would not result in a detectible impact on benthic biota (e.g. a reduction in biomass). These areas can be large, but at any point in time the dredge plumes are likely to be restricted to a relatively small portion of the ZOI (see Figure 1). The outer boundary of the ZOI bounds the composite of all of the predicted maximum extents of dredge plumes and represents the point beyond which dredge-generated plumes should not be discernible from background conditions at any stage during the dredging campaign. Furthermore, this provides transparency for the public regarding where visible plumes may be present, albeit only occasionally, if the proposal is implemented. Reference sites for monitoring natural variability would ideally be located outside of the ZOI.

Predictions of both impacts to, and serious damage/irreversible losses of, benthic communities and habitats (i.e. the ZOMI and ZOHI) should be considered and presented in the context of the *Technical Guidance – Protection of benthic communities and habitats* (EPA 2016a).

### 3.4.2 Presenting the zonation scheme

The system of zones is designed to be presented in a spatially-based form. Figure 1 shows a zoomed-out view of how the zonation scheme would be represented. It shows the relative sizes of the zones that are likely to be generated based on recent experiences and also shows that all effects of dredging should be captured by the outer boundary of the ZOI.

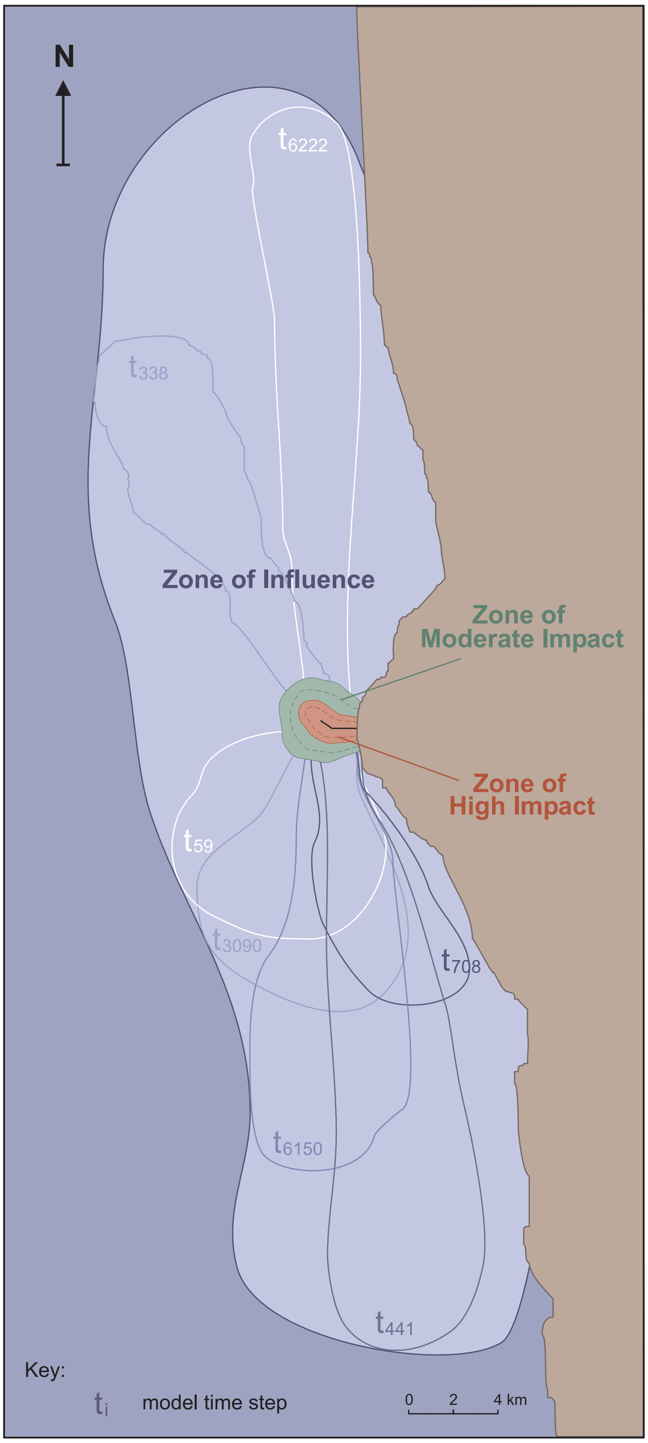


**Figure 1: A schematic representation of the spatially-based zonation scheme for representing dredging related impacts where red represents the Zone of High Impact, green represents the Zone of Moderate Impact and pale blue represents the Zone of Influence. The outer boundaries of individual dredge plumes are shown as blue shaded lines within the Zone of Influence at different time steps ($t_n$) during a simulated dredging campaign.**

In simple terms, the level of cumulative pressure on biota from dredge-generated sediments will generally decrease with distance from the dredging site. As a result, the degree of impact would similarly be expected to decrease with distance from the dredge site. Figure 2 shows how the pressure and resultant degree of impact on benthic communities would change with distance from dredging, and how these changes can be represented by the zonation scheme described above.

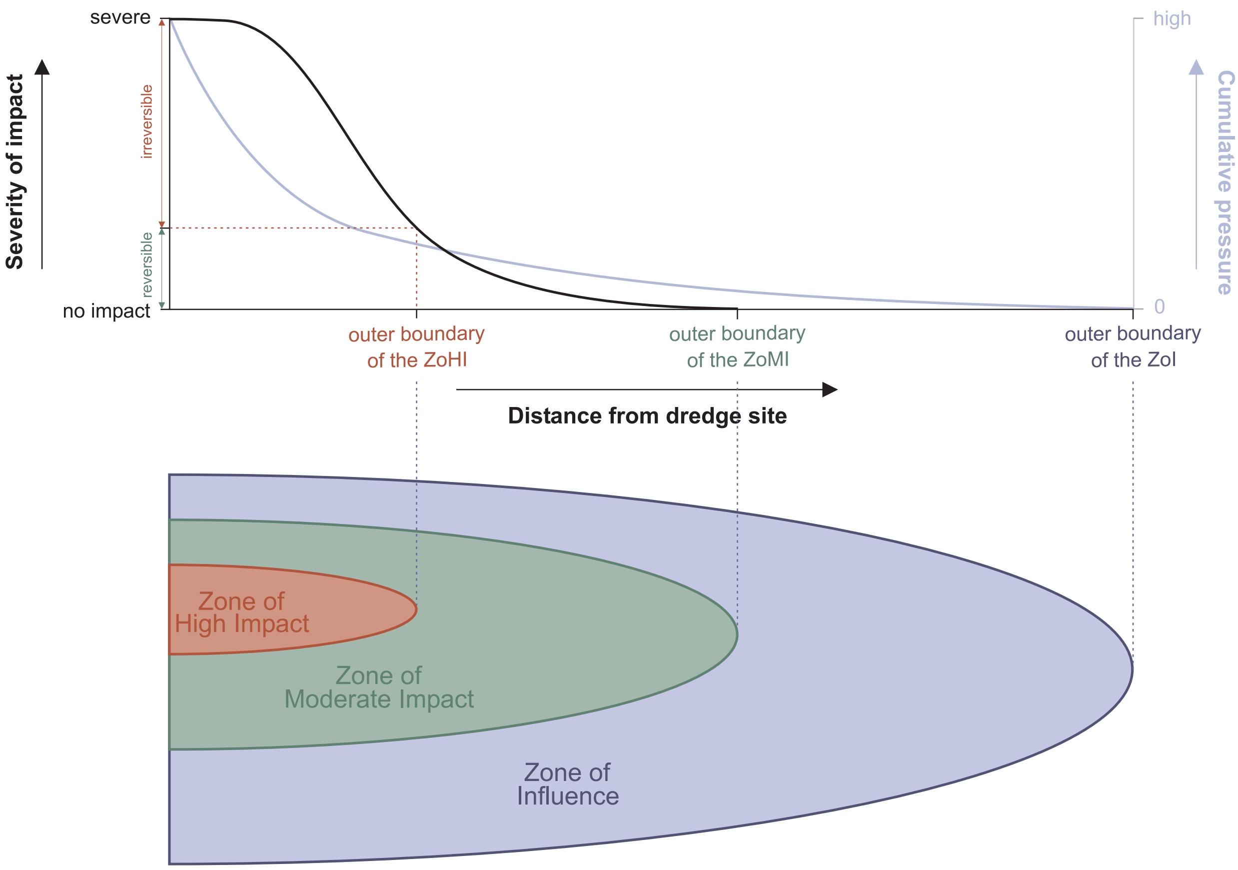


**Figure 2: A schematic representation of the degree of change in environmental quality associated with dredging (grey line) and level of resultant impact to benthic communities (black line) along a transect extending away from the dredging site to the outer extremity of the Zone of Influence. The location of the outer boundaries of the Zone of High Impact (ZOHI), Zone of Moderate Impact (ZOMI) and Zone of Influence (ZOI) are shown relative to these predicted changes in environmental quality and impacts on biota.**

The level of pressure and resultant ecological impact associated with dredging would generally be expected to attenuate with distance from the dredge site as represented by the black line in the upper panel of Figure 2. This figure also shows the position of the outer boundaries of the Zones of High and Moderate Impact relative to the level of impact expressed here as 'reversibility'. A key point to note is that all impacts relevant to a particular zone are attenuated within that zone before transition into the next zone further from the source of suspended sediments. For the ZOHI, this means that no serious/irreversible impacts should be predicted to occur outside of this zone and the corollary is that not all impacts on all biota within this zone are predicted to be serious or irreversible. Near to the boundary with the ZOMI, but still within the ZOHI, the level of impact can logically be expected to be lower than closer to the dredge site and approaching the point where there are no serious/irreversible impacts. Most importantly there should be no serious/irreversible impacts on benthic communities in the ZOMI or beyond.

Similarly, moving further along the transect away from the dredging site a point would be reached near the ZOI but still within the ZOMI where there would be practically no detectible impact on biota. A suite of guidelines that could be used to assist in predicting the spatial extent of these zones is provided in Appendix A.

> The spatially-based zonation scheme provides a clear and consistent way of describing and presenting the extent, severity and duration of predicted impacts of dredging for EIA.

### 3.4.3 Accounting for predictive uncertainty

Uncertainty is a factor inherent in all predictions and there is an array of sources of uncertainty associated with dredging impact predictions. In order to take account of this uncertainty in the EIA process, the final set of predictions may describe the lower and upper ends of the *likely* range of impacts associated with the proposal (i.e. the likely 'best-case' and the likely 'worst-case'). This range should be *realistic* and based on understanding of probable scenarios and their associated environmental outcomes. It should not include *unrealistic* best-case or worst-case (or other improbable) predictions.

This is illustrated conceptually in Figure 3, which shows the likely location of the outer boundaries of the high and moderate impact zones along a transect extending away from the dredging site. The transect line at the bottom of the figure has two sections marked 'likely range', which represent the range of possible positions of the boundary of each zone. The distances from the dredging site that correspond to the two ends of each marked section represent the likely 'best-case' and likely 'worst-case' positions of that boundary.

> In order to take account of this uncertainty in the EIA process, the final set of predictions should describe the lower (*likely* best-case) and upper (*likely* worst-case) ends of the likely range of positions of the boundary that could reasonably be expected based on understanding of probable scenarios and their associated environmental outcomes.

The pair of boundaries might be generated using a number of different approaches, but in practice the process will always involve predicting pressure through dredge plume modelling and assessing impacts by interrogating the pressure fields against biological effects criteria.

Dredge plume modelling will need to consider variability in physical forcings (e.g. typical and atypical wind conditions), sediment release rates (e.g. more fines, less fines), and dredge operation and management scenarios (e.g. different dredge types and operating modes). Biological-effects thresholds will need to account for the tolerance and susceptibility of different species and groups to the same level of sediment-related pressure and give particular attention to the most sensitive groups of benthic organisms or community/habitat types.

In all cases there will need to be a degree of 'professional judgement' employed to establish the likely best-case and likely worst-case locations of the boundary for a zone.

Section 3.5. provides more detailed explanation of this process and the associated technical considerations by way of simple examples.

> The range of likely impact predictions should be based on the best available design, construction and management techniques and approaches being applied to dredging and its management.

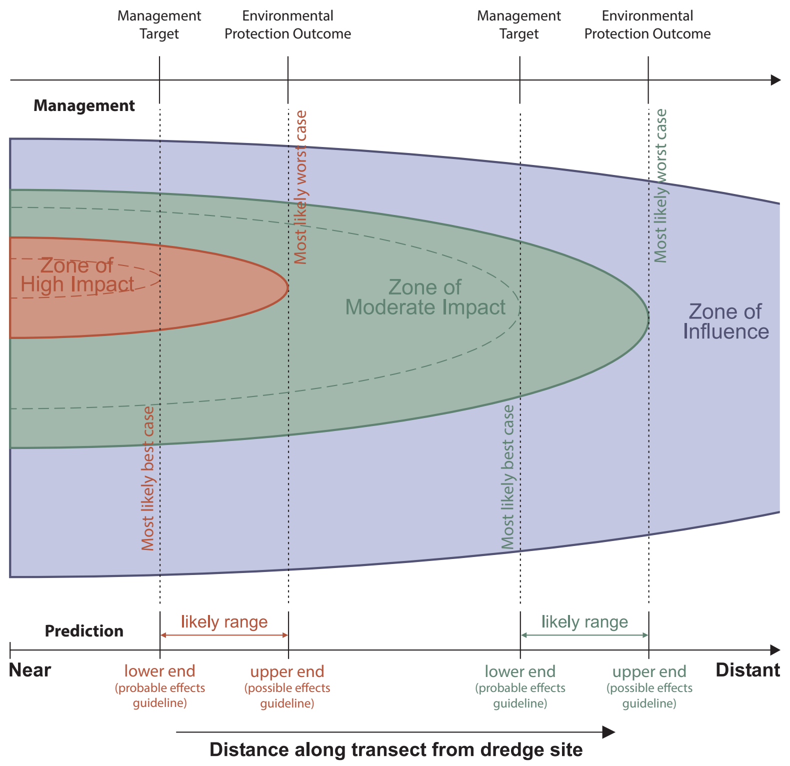


**Figure 3: A conceptual representation of the 'likely' range of realistic locations for boundaries of the ZOHI and ZOMI associated with a dredging proposal and how this is translated into the spatial zonation scheme for presenting impacts for environmental impact assessment.**

### 3.4.4 Presenting realistic and likely predicted impacts

Boundaries that represent the range of likely environmental impacts should be presented in map form and overlay the benthic habitat map as shown in Figure 4. Figure 4 (a) shows the full extent of the predicted ZOI and the ZOHI and ZOMI within it. Figure 4 (b) shows boundaries associated with the ZOHI and ZOMI, where the broken and solid lines represent the likely best-case and likely worst-case respectively.

In making and presenting predictions in the manner shown in Figure 4, proponents should consider the likely best-case as reflecting a Management Target they are hopeful of achieving if all goes well and all reasonable and practicable measures to minimise or avoid impacts are applied to dredging and its management. The likely worst-case on the other hand would reflect an Environmental Protection Outcome that the proponent is *confident* of achieving using all reasonable and practicable measures even if things do not go as well as hoped.

These maps serve a number of key purposes. Firstly, they present fundamental information for effective EIA, including information about the extent, severity and duration of predicted impacts, and the full extent of the predicted ZOI, which ensures there is a common basis for understanding the potential extent of sediment plumes anticipated during the dredging operations and resultant impacts. These maps explain predictive uncertainty and clearly differentiate between the targets that the proponent will aim for, and the outcomes that they are confident in achieving, through management of the project.

Proponents will be expected to consider the range of likely impacts when developing their proposed environmental monitoring and management strategies.

> The lower end of the range of likely impacts should reflect a likely best-case outcome that would become a target for management. The upper end of the range should reflect a likely worst-case outcome that the proponent is both confident of achieving and prepared to be conditioned to.

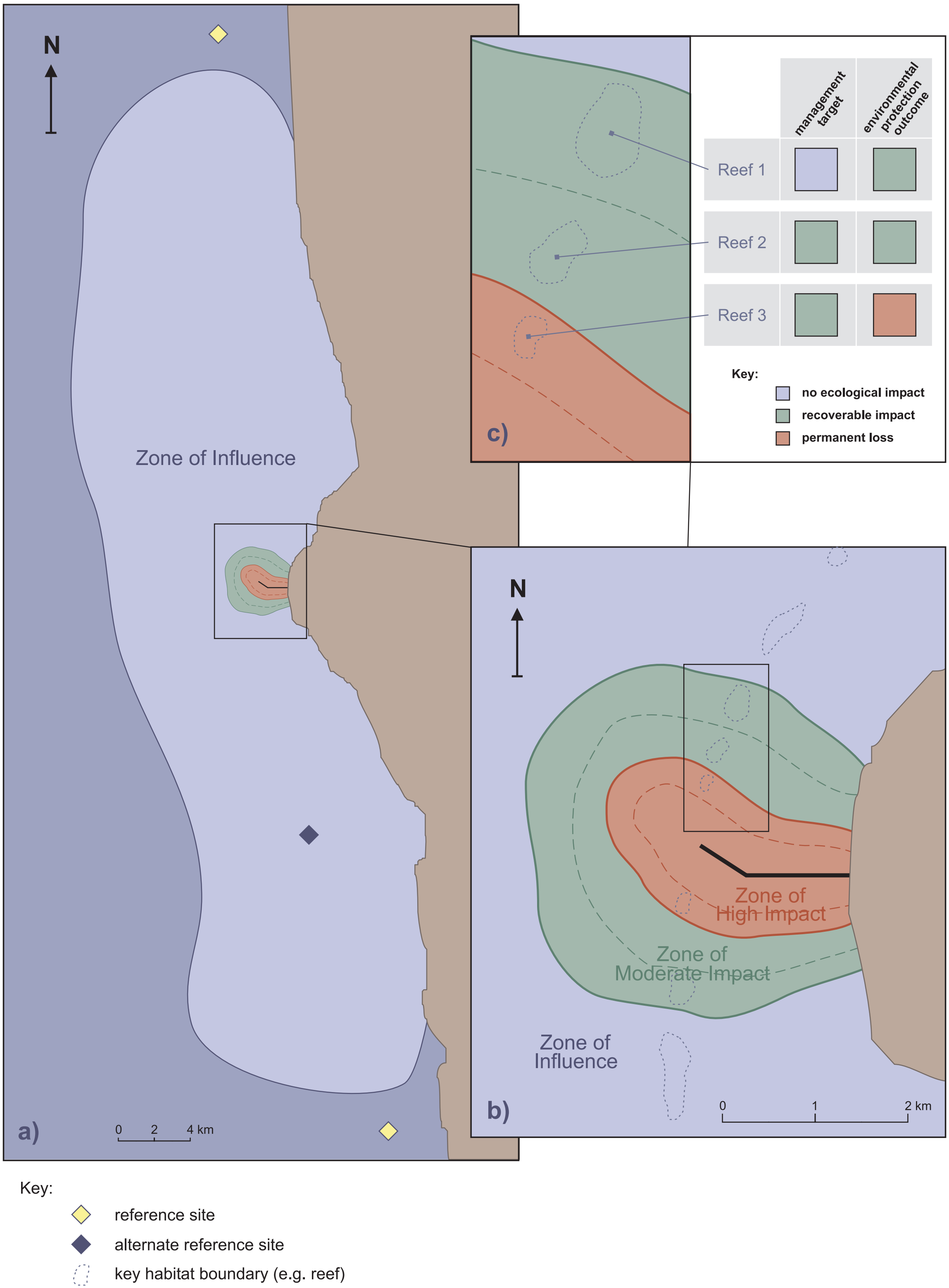


**Figure 4: An example map-form presentation of: a) the predicted ZOI, ZOMI and ZOHI associated with channel dredging (represented by the black line), b) closer view of the predicted ZOHI and ZOMI, noting that the area between the broken lines (inner) and solid lines (outer) represents the uncertainty associated with the location of the zone boundary, and c) zoomed in section showing the management targets and expected environmental outcomes for the zones and the area of uncertainty within the zones.**

## 3.5 Impact prediction

Notwithstanding the inherent uncertainties associated with predicting realistic and likely dredging impacts, judgements will ultimately need to be made on where to locate the boundaries that spatially define the outer extents of the ZOMI and ZOHI. As explained above in section 3.4, each zone will have two boundaries; one to represent where the Environmental Protection Outcome (EPO) will be met and the other where the Management Target (MT) is located (see Figures 3 and 4). They are based on predicting the pressure fields associated with dredging and linking this to knowledge of the susceptibility and resilience of benthic species to dredging pressures.

There are two basic strategies that could be used to define the likely 'best-case' and likely 'worst-case' zone boundaries for MT and EPO purposes: i) interrogating two different sets of pressure field data against a single biological effects criterion, or ii) interrogating a single pressure field dataset against two different sets of biological effects guidelines. These two approaches are best explained by way of simple examples.

### 3.5.1 Using different pressure fields

Consider an example where a particular species is tolerant of SSC up to 10 mg/L (i.e. it showed sub-lethal effects when exposed to >10 mg/L SSC). In this case the outputs of the sediment transport and fate model could be interrogated over the time-course of dredging to identify model cells that experienced more than 10 mg/L at any point during the dredging and cells that did not. In this circumstance the contour line between the >10 mg/L and ≤10 mg/L would provide a basis to delineate between effect and no-effect and as such could be used to locate the outer boundary of the ZOMI. This is essentially taking the highest percentile (100$^{th}$ percentile) of the predicted SSC; an approach that is highly conservative (and probably unrealistic).

An alternative approach might be to search for cells that are predicted to experience 10 mg/L for 5% or more of the time (i.e. the 95$^{th}$ percentile of the 10 mg/L concentration). This will tend to drive the boundary closer to the source of turbidity but remains very conservative and more representative of a 'realistic worst-case scenario' where effects are 'possible'. This would be a reasonable approach for determining the EPO boundary.

If, on the other hand, the *median* value for each cell was compared against the 10 mg/L criterion, the contour line would be much closer to the turbidity source. The cells on the contour line are predicted to have experienced more than 10 mg/L for half the time and less that 10 mg/L for half the time. This line would represent a more 'average' or most likely best-case where effects are more 'probable' if the guideline is not met, and would therefore be a rational basis to delineate the MT boundary between the ZOMI and ZOI.

Similar approaches could be used to determine the boundary between the ZOMI and the ZOHI by replacing the criterion for a pressure intensity that caused sub-lethal effects with one that caused mortality. Sun et al (2019) provides guidance on dredging-related sediment transport modelling and incorporation of uncertainty.

### 3.5.2 Using different biological effects guidelines

In the example above the pressure criterion (i.e. 10 mg/L) was kept constant and the 'uncertainty' was captured by interrogating the model against the 95$^{th}$ and 50$^{th}$ percentiles to determine the EPO and MT boundaries respectively. An alternative approach is to interrogate a single set of model outputs against two biological effects criteria; one that represents a 'conservative' threshold for determining the EPO and one that represents a 'less conservative' threshold for determining the MT boundary.

For example, consider a case where the 10 mg/L guideline was derived from experiments where exposure to SSC of 10 mg/L caused sub-lethal effects in the most sensitive species tested, but in all other species tested showed no effects until SSC reached 20 mg/L. When interpreting these results in the context of predicting impacts, it might be reasonable to conclude that effects are 'possible' where SSC reached 10 mg/L and effects are 'probable' where SSC reached 20 mg/L. Using this as an example, a single sediment transport model output could be interrogated against the 'possible-effects' guideline (i.e. 95th percentile of the 10 mg/L concentration) and 'probable-effects' guideline (i.e. the 95$^{th}$ percentile of the 20 mg/L concentration) to determine the location of the EPO and MT boundaries respectively (see Figure 3).

### 3.5.3 Applying biological effects guidelines

The biological 'effects' criteria in the example above are very simplistic and in practice biological effects criteria will also have a frequency of occurrence and/or duration of occurrence (e.g. >10 mg/L for ≤ 5 consecutive days). Furthermore the 'pressure' of relevance to the benthic biota might not be SSC directly; it is more likely to be related to the resultant benthic light availability and/or sediment deposition rate.

Sediment deposition rate is complicated to predict and measure in ecologically-relevant terms (see Jones et al 2019a). Furthermore, high levels of sediment deposition are generally localised and associated with very high SSC in areas close to sediment excavation and disposal sites.

Benthic light availability on the other hand is more readily measured in the field and dredging-related effects on benthic light availability can be widespread. Phototrophic benthic primary producers require sufficient light to survive and many light-related guidelines have been developed from the WAMSI DSN research to assist in predicting and managing dredging related impacts and these are presented in Appendix A.

To apply these light-related guidelines at the EIA stage, the outputs of sediment transport models (e.g. SSC) would need to be transformed to light availability as a function of depth to allow the mean daily light received at the seabed to be calculated and expressed as mol photons $m^{-2}$ $d^{-1}$. It is important that background levels of turbidity and associated light attenuation are also accounted for (e.g. in the model or empirically) to provide likely actual light at the seabed (i.e. light attenuation due to natural factors plus the additional attenuation due to dredging-related turbidity).

Model outputs for bottom light at each grid cell can then be interrogated against the guideline values for the species present at the location to determine the likelihood of measurable impacts, and boundaries drawn accordingly.

### 3.5.4 Deriving biological effects guidelines

The pressures generated by dredging are interrelated in that there will always be lower light availability when suspended sediment concentrations are high, so it is difficult to disentangle the relative importance of the different impact pathways based solely on the results of in-situ monitoring programs.

The WAMSI DSN systematically explored a number of these interrelated factors under laboratory conditions to determine the relative importance of each pressure on the key functional groups of benthic organisms found in WA: corals, seagrasses and sponges. The species used in the studies were selected so that the results could be more generally transferable and applied to other species and genera with similar morphologies, physiologies or growth habits where information on those taxa did not exist.

The majority of species selected for study have broad biogeographic ranges, occurring in the Pilbara and other parts of northern Australia (including the Great Barrier Reef) and the Indo-Pacific. The distribution of at least one species of coral, seagrass and sponge extends to the south coast of WA.

The results of the WAMSI DSN research, coupled with professional judgement, have been used where possible to derive a set of pragmatic and relatively high confidence guideline values that could be used to delineate the outer (i.e. worst-case) boundaries of the ZOHI and ZOMI for EPO purposes. For the purposes of this guidance these are termed 'possible-effects' guidelines. A set of less conservative (but still reasonable and plausible) guidelines were also developed that could be used to delineate the inner (i.e. best-case) ZOHI and ZOMI boundaries for MT purposes, and these are termed 'probable-effects' guidelines.

'Possible-effects' guidelines are very conservative (worst-case) and would be used to determine the location of a zone boundary for EPO purposes. 'Probable-effects' guidelines are less conservative (best-case) and would be used to determine a zone boundary for MT purposes.

The relevant key findings and suggested 'possible-effects' and 'probable-effects' guideline values for corals, seagrasses and sponges are presented in Appendix A. Proponents should consider these values as a guide only and use WAMSI DSN research, other relevant scientific research and professional judgement to determine the most defendable values for their proposed dredging project. The final values chosen to predict the boundaries of the three different zones, and to monitor and manage for potential impacts to benthic communities and habitat (BCH), is the responsibility of the proponent and will be influenced by the nature and scale of the proposed activities, the regional location, resident BCH and background water quality conditions.

## 3.6 Integrating predictions with monitoring and management

In an ideal world, predictions would be 100% accurate, and this would facilitate straightforward EIA and reduce or negate the need for monitoring and reactive management. The WAMSI DSN and the associated scientific reports and papers have significantly improved our understanding of conditions likely to cause lethal and sub-lethal stress on BCH during dredging projects. However, dependent on the location, nature and scale of dredging, a range of environmental monitoring and management strategies may be required to ensure that impacts are minimised during project implementation and to demonstrate compliance with any limits established through the approval process.

By presenting predictions that represent the lower and upper ends of the range of likely impacts, the framework establishes a logical and consistent basis for translating those predictions into monitoring and management strategies and conditions of approval. The likely best-case predictions will be used in setting appropriate 'management' objectives (i.e. targets) whilst the likely worst-case predictions would be more aligned with environmental protection outcomes (i.e. regulatory limits). Importantly this allows a distinction to be made between monitoring requirements for informing management of dredge operations and monitoring requirements for demonstrating compliance with Ministerial Conditions of approval. This should allow a more efficient allocation of resources between the various monitoring and management tasks.

In simple terms, proponents can expect that the frequency and extent of compliance monitoring during the dredging programs will be inversely proportional to the overall confidence in the predictions of environmental impact. The environmental setting and the significance of the potential and likely impacts, and the effectiveness and responsiveness of the proposed environmental monitoring and management strategies, will also be considerations.

Developing the detail around proposed monitoring to inform adaptive management and determine if management targets are being achieved would generally be a task for proponents. However, there may be cases where, based on its consideration of information provided for assessment, the EPA will make recommendations in this regard. When developing proposed environmental monitoring programs, both adaptive management monitoring and compliance monitoring should be considered and there may be efficiencies that could be realised by running the programs concurrently.

Proponents could expect the highest monitoring and management burden in situations where environmental values are high and where there are high levels of predictive uncertainty.

Monitoring and adaptive management in the various zones will have differing objectives. The environmental significance of the area and the level of predictive uncertainty exposed during EIA will inform how much monitoring is required.

In addition to minimising impacts of dredging on benthic habitats and communities, an overarching objective of the assessment framework, outlined in the preceding sections, is to enhance the linkage between the environmental impact predictions made for EIA and the data generated through monitoring and management programs implemented post-approval. This should generate validation data that will further increase confidence over the prediction – management continuum.

As the knowledge generated from more targeted monitoring of dredging pressures and impacts is applied in new EIA, confidence in dredging-related impact predictions should increase allowing monitoring requirements to be reduced over time without reducing overall confidence.

The EPA strongly supports greater public availability of environmental data collected for EIA and post-approval monitoring and management programs and may recommend conditions to facilitate this outcome. The information already provided can be accessed and shared through the Index of Marine Surveys for Assessments (IMSA), an online portal to information about marine-based environmental surveys for EIA in WA established and maintained by the Department of Water and Environmental Regulation.

### 3.6.1 Dredging environmental monitoring and management plans

The fundamental purposes of a dredging environmental monitoring and management plan (DEMMP) are to minimise impact and ensure that the environmental protection outcomes established for a project are not compromised. The proponent should also consider structuring the DEMMP so that the monitoring data are able to inform adaptive management of the dredging program to minimise the impacts and achieve the relevant management targets. As such, the DEMMP should focus on the key threats posed by the project and the pathways by which those threats could cause the environmental protection outcomes to be compromised. The primary threats to the surrounding marine environment from dredge-generated sediment are shading caused by sediments suspended in the water column and smothering of benthic habitats and organisms caused by the deposition of these sediments.

The DEMMP should be designed to achieve management targets that indicate a level of impact that is lower than the limits established as environmental protection outcomes. As such, the DEMMP is designed to provide early warning of adverse trends and trigger pre-emptive management before the required environmental protection outcomes are compromised. The DEMMP should also be designed to monitor and report on the important pressures generated by the dredging campaign so that any observed impacts can be attributed to the project and the impact prediction models can be validated and fine-tuned through improved understand of the cause/effect relationships.

Environmental monitoring and management plans should be structured so that a focus on achieving the management targets would provide a high degree of confidence that the environmental protection outcomes are not compromised.

A DEMMP should be clear and unambiguous and contain the following key elements:

- clearly stated objectives
- a monitoring/management feedback loop to achieve those objectives
- management triggers along pressure-response pathways
- monitoring regime including site locations and methods to provide data to allow assessment against the management triggers
- clearly set out data evaluation procedures to identify where and when management triggers have been reached
- contingency management strategies to be employed if triggers are reached
- a reporting process.

The EPA expects the most relevant scientific information to be used when preparing a DEMMP which may require proponents to undertake pre-referral baseline monitoring to provide the necessary local context. Proponents should provide the DEMMP as part of the documentation submitted for assessment. These plans should contain sufficient information to allow the monitoring methods, data interpretation and the efficacy of proposed management to be assessed.

> Dredging environmental monitoring and management plans are an integral part of the documentation submitted for EIA of dredging proposals.

### 3.6.2 Environmental monitoring locations and their purposes

Selection of locations for establishing monitoring and reference sites should be based on a number of considerations including the locations of predicted zone boundaries (including the area of uncertainty), the types and locations of benthic communities in those zones and to provide early warning of potential impacts to the different benthic communities.

For example, because the ZOHI is based on the extent of essentially irreversible impacts and any approval that might be granted would recognise that, it would not be necessary to monitor the health of benthic communities in that zone for 'compliance' purposes. There would however, be significant benefit from monitoring both dredge-related 'pressure' and 'ecological response' along a gradient from near the dredging location through to the edge of this zone (and beyond). In the short term, the results of pressure and response monitoring would help to appraise and refine some of the early warning trigger criteria used for 'management' of impacts in the ZOMI/ZOI during the course of the dredging campaign (i.e. adaptive management). In the longer term, benefits would be realised through improved understanding to inform assessments of future proposals for new capital or maintenance dredging.

The ZOMI is a key focus for monitoring and management as this is the transition zone between where permanent loss and no effects are predicted. Monitoring and management in the ZOMI serves dual purposes to 1) minimise impacts though informed adaptive management designed to at least achieve a management target, and 2) ensure any impacts that do occur are reversible and not greater than approved (i.e. consistent with the environmental protection outcomes prescribed in the Ministerial Conditions of approval). In this zone it would be expected that monitoring would include both dredge-related 'pressure' and 'ecological response' in consideration of a risk-based environmental monitoring and management framework (section 3.6.3).

The overarching objective of monitoring and management in the ZOI is to ensure there are no detectible effects of dredging on benthic communities in that zone.

As a rule, monitoring locations for a zone should be as close as possible to the inner boundary for that zone. This is particularly important for the ZOI, given its size, and so these monitoring locations should be established in suitable habitats as close to the ZOI/ZOMI boundary as possible.

Reference sites should be located outside of the predicted ZOI (see Figure 4). However, given the potential scale of the ZOI, it may prove to be logistically very difficult to establish and regularly monitor sites that are very distant from the central area of activity. Furthermore, the environmental conditions outside of the ZOI may be such that there are few appropriate areas that have the necessary degree of similarity to the impact monitoring sites to be appropriate as reference sites. In acknowledgement of these issues, the EPA will consider operational reference sites within the ZOI, if well justified and where it can be demonstrated that the frequency and intensity of exposure to dredging plumes is low. Notwithstanding the above, the EPA would still expect reference sites to be established outside of the ZOI as a safety measure, but would accept a lower monitoring frequency than at the operational reference sites.

> Reference sites should ideally be established outside the Zone of Influence but monitoring regimes that include reference sites within this zone may be considered if well justified.

### 3.6.3 A risk-based environmental monitoring and management framework

The framework around which to design environmental monitoring programs should be risk-based and incorporate the best scientific understanding of pressure-response pathways for key biota in the benthic communities to be monitored. Essentially this means that monitoring would be designed around indicators that signify progressively greater risk of unacceptable impact and should reflect the location, nature and scale of the dredging program. Monitoring may take the following general risk-based form and apply suitable techniques to measure the responses in primary, secondary and tertiary indicators as set out below.

1. **Primary indicators** signify a very early warning of potential threat and low level of risk to the biota of interest. A primary indicator could be a measure directly linked to a pressure from dredging such as turbidity, light attenuation coefficient/benthic daily light integral or sediment deposition rate. Exceeding a guideline linked to a primary indicator would trigger tier 1 management, which could include ***investigating the cause of the exceedance*** and ***increasing monitoring*** to include a secondary indicator.
2. **Secondary indicators** signify a moderate risk to the biota of interest and might include measures of biotic stress such as change in the colour of coral tissues or a reduction in the shoot density of seagrass. Exceeding a guideline linked to a secondary indicator would trigger tier 2 adaptive management, which could include implementation of measures to ***reduce dredge-related pressure*** and monitoring of a tertiary indicator.
3. **Tertiary indicators** signify a high and unacceptable level of risk to the biota of interest. A tertiary indicator would be a measure or measures that are immediate pre-cursors to an unacceptable impact. Exceeding guidelines linked to a tertiary indicator would trigger strong management action to ***alleviate pressure before unacceptable impacts occur.***

> Dredging environmental monitoring and management plans should reflect contemporary best available techniques and approaches and ideally be risk-based, using readily measurable indicators along the pressure-response pathway, to trigger management to prevent unacceptable impacts.

An objective of the integrated EIA and environmental monitoring and management approach is to provide for an explicit description of environmental impacts and outcomes of dredging. Proponents should therefore expect that the EPA may incorporate the predicted zone boundaries into conditions it may recommend to the Minister for Environment.

The clear definition of project impacts (in terms of extent, severity and duration) and areas to be protected allows for unambiguous audit of project performance against approval conditions, which in turn reduces uncertainty around compliance or enforcement issues.

The strong links between predictions, approvals and associated management requirements highlight the importance of robust model calibration and validation, and high-quality science - all targeted towards reducing uncertainty in both prediction and management. Appendix C provides advice to assist proponents to develop and implement their monitoring and management plans efficiently and effectively.

### 3.6.4 Using 'possible-effects' and 'probable-effects' guidelines for monitoring and management

A number of 'possible-effects' and 'probable-effects' guidelines for key benthic species and communities have been presented in Appendix A. They are designed to be of use for establishing the spatial extent of 'limit' of acceptable impacts (delineated by the EPO boundary) and also the spatial extent that the proponent will endeavour to contain impacts within (delineated by the MT boundary). These guidelines are 'generic values', based on knowledge of pressure-response relationships and are suitable for use in the absence of more site-specific guidelines, noting however that proponents are encouraged to develop an appropriate set of guidelines for their proposed dredging project.

For any pair of 'possible-effects' and 'probable-effects' guidelines, the 'probable-effects' guidelines will reflect a higher pressure than the 'possible-effects' guidelines. When considered in the context of a *risk-based* management framework, and focussing on *meeting the management target*, it would not be advisable to apply the 'probable-effects' guidelines used to derive the location of the boundary as primary or early warning (i.e. Tier 1) indicators to 'manage' at that location. That is because the objective of management is to prevent unacceptable effects and, by definition, effects are probable if pressure reaches that point. It would be more sensible to use the 'possible-effects' guidelines (used to locate the EPO boundary) as early warning indicators at the MT boundary, because if reached would only signify a 'possibility' of an effect. Importantly it would also indicate that the pressure at the EPO boundary would be below the 'possible effects' guidelines given the attenuation of pressure with distance from the turbidity source.

Using the same logic and approach, the possible and probable effects guidelines could also be used at the EPO boundary for compliance monitoring. But in this situation, and using the ZOHI/ZOMI EPO boundary as an example, the possible effects guidelines for ZOMI/ZOI boundary could be used as early warning indicators as they would be triggered at a lower level of pressure - where sub-lethal effects are just 'possible'.

Where guidelines contain threshold (e.g. >2 DLI) and/or duration components (e.g. must be met for any 14-day period) it is not unreasonable to arbitrarily set early warning indicators that signify less pressure than the threshold (e.g. >3 DLI instead of >2 DLI) and/or for shorter duration (e.g. 7 days instead of 14 days). Professional judgement will need to be applied to ensure the risk-based management framework is logical, practical to implement and fit-for-purpose.

Further advice and suggested approaches and guidelines for tiered management that link back to the DSN research findings are provided in Appendix C.

## 3.7 Critical windows of environmental sensitivity

When designing dredging proposals and making predictions of environmental impacts, proponents should consider *critical windows of environmental sensitivity*. Critical windows of environmental sensitivity include times of the year or particular sites where key species, ecological communities or critical processes may be particularly vulnerable to pressures from dredging.

There are numerous examples of known critical windows of marine environmental sensitivity. Some examples which the EPA has either considered previously in relation to dredging proposals, or is aware of, include spawning and larval settlement periods for corals, habitat for spawning aggregations and juveniles of fish (e.g. pink snapper) and invertebrates (e.g. blue swimmer crabs), critical habitat for breeding of marine wildlife (e.g. turtles, dugong), the timing and routes for migration of specially-protected migratory species (e.g. JAMBA/CAMBA listed migratory birds and whales) and habitat that supports primary food resources for threatened marine fauna listed under State and Commonwealth legislation (e.g. seagrass areas in Shark Bay and Exmouth Gulf grazed by dugong).

Knowledge of critical windows of environmental sensitivity provides an opportunity to develop avoidance strategies to reduce risk to marine communities (see section 3.8) and contribute to impact avoidance/mitigation as set out in section 3.1.

The WAMSI DSN undertook research to identify and collate relevant information to better understand and manage the potential impacts of dredging on critical life cycle processes of important marine taxa in WA. In addition, significant effort was put into identifying and quantifying key dredging-related pressure:response pathways that could affect coral reproduction, settlement and survival. The key findings of the research on corals, fishes and other taxa are presented in Appendix B.

## 3.8 Contemporary construction, design and management approaches for minimising impacts of dredging

While the best and most appropriate measures to avoid or minimise dredging related impacts tend to be highly site and project specific, some examples include:

- Up-front design to minimise the need for dredging, considering the environmental setting and operational requirements.
- Dredge area design that aims to minimise direct and indirect impacts on key benthic habitats (e.g. design and locate marine infrastructure to avoid or reduce impacts on coral or algal reefs, seagrass and filter feeder habitats or mangroves).
- Using site-specific geotechnical data and understanding of dredge equipment-substrate interactions to help select *fit for purpose* dredging equipment and operating modes to minimise the environmental impacts.
- Using this knowledge of geotechnical conditions, and dredge equipment-substrate interactions to establish the likely physical characteristics and generation rates of fines produced by dredging at the site.
- Using validated hydrodynamic and sediment transport models to assess the dynamics and likely fate of sediment plumes.
- The use of physical interventions such as silt curtains, where they are operable and likely to be effective in controlling turbidity release and dispersion.
- Contracting dredges equipped with sediment management devices where these are found to minimise sediment generation and dispersion.

- Scheduling dredging to account for, and avoid, periods or areas of environmental sensitivity such as coral spawning and turtle nesting. Particularly useful for short-duration dredging campaigns.
- A commitment to manage dredging in ways that minimise the release of sediments into the water column as much as practicable, particularly in situations where dredging-related sediments have the potential to impact sediment-sensitive benthic communities. Methodologies such as 'no overflow' or 'planned commencement of overflow', piping dredge spoil direct to disposal sites or to transfer vessels stationed sufficient distances from sensitive receptors to eliminate or minimise risk pathways to those receptors, may need to be considered.
- The application of near real-time data collection and interpretation methods (particularly for turbidity) to support environmental management of dredging. This should be determined on a hierarchical basis grading from small maintenance dredging campaigns in low sensitivity environments where near real-time monitoring is not warranted through to major capital dredging projects where substantial commitments to monitoring and adaptive management, including the use of telemetered water quality instruments, are required. In addition to the scale and environmental settings of proposals, in all cases the degree of uncertainty in impact prediction will be considered when determining the appropriate level of near real-time data collection and interpretation required to manage project implementation.
- Stopping or delaying dredging where unanticipated and significant dredging related impacts cannot be avoided or managed and have, or are likely to, result in exceedance of an EPO.

# 4 Definitions

| Word, phrase or acronym | Definition for the purpose of this Technical Guidance |
|---|---|
| **BACI** | Before/After and Control/Impact |
| **BCH** | Benthic communities and habitats |
| **Best-case** | See Likely best-case |
| **Bioindicators** | Biological aspects of the environment that respond to stressors, such as suspended sediments, in a known way. They provide direct measures of sub-lethal effects or impacts and are often used in environmental monitoring programs to assess ecological health. |
| **CCA** | Crustose coraline algae |
| **CDOM** | Coloured dissolved organic matter |
| **DEMMP** | Dredging environmental monitoring and management plan |
| **DLI** | Daily light integral; the cumulative amount of light received during daylight hours |
| **DPIRD** | Department of Primary Industries and Regional Development |
| **Dredge spoil** | Seabed substrate material after it has been excavated from the seabed |
| **Dredging** | Involves excavation of the seabed from the upper intertidal zone to the subtidal zone. Dredging in the sense of this guidance means both dredging and dredge spoil disposal activities. |
| **DSN** | Dredging Science Node; a scientific program undertaken by WAMSI and designed to assist in the prediction and management of environmental impacts associated with marine dredging programs in Western Australia and more broadly. |
| **DWER** | Department of Water and Environmental Regulation |
| **EIA** | Environmental impact assessment |
| **EIAM** | Environmental impact assessment and management |
| **EPA** | Environmental Protection Authority of Western Australia |
| **EPO** | Environmental Protection Outcome. A level of impact that equates to the likely worst-case and designed to be reflected as an impact limit in any conditions of approval. In that case proponents would be required to demonstrate compliance with the EPO. |
| **EPS** | Extracellular polymeric substances |
| **$ETR_{MAX}$** | Maximum electron transport rate; a measure of photosynthesis |
| **Extent** | The area over which an impact extends |
| **Functional groups** | Groups of species (which are not necessarily related generically) that share similar important ecological characteristics and play equivalent roles in the functioning of the biological community. |

| Word, phrase or acronym | Definition for the purpose of this Technical Guidance |
|---|---|
| **IMSA** | The Index of Marine Surveys for Assessment (IMSA) is an online portal for the systematic capture and sharing of marine data created as part of EIA and administered by DWER. |
| **Infrastructure** | Shipping channels, turning basins, berth pockets, pipeline trenches, spoil disposal sites, sub-sea mine areas and land reclamations are some examples of infrastructure. |
| **Irreversible** | Lacking a capacity to return or recover to a state resembling that prior to being impacted (also see reversible). |
| **JAMBA/CAMBA** | Bilateral migratory bird agreements between Australia and Japan (JAMBA), and Australia and China (CAMBA). |
| **LAC** | Light attenuation coefficient; a measure of water clarity. |
| **LAU** | Local Assessment Unit for assessing cumulative impacts to benthic communities and habitats (see EPA 2016a). |
| **Likely best-case** | Level of impact that represents the lower end of the likely range. |
| **Likely worst-case** | Level of impact that represents the upper end of the likely range. |
| **MODIS** | Satellite-based Moderate Resolution Imaging Spectroradiometer. Terra MODIS and Aqua MODIS are acquiring data in 36 spectral bands covering the entire Earth's surface every 1 to 2 days. |
| **MT** | Management Target. A level of impact that equates to the likely best-case and is used as a target for management. Proponents would be expected to design and implement monitoring and management plans that aim to meet the MT and ensure compliance with the EPO. |
| **Near real-time** | Refers to a system for monitoring and interpreting data where the time lag between collecting monitoring data and responding is sufficiently short to be considered as immediate as practicable. |
| **NOEC** | No observable effects concentration |
| **NTU** | Nephelometric Turbidity Units |
| **NW WA** | North-west Western Australia |
| **OM** | Organic matter |
| **OTUs** | Operational Taxonomic Units |
| **PAR** | Photosynthetically Active Radiation |
| **Persistence** | The period of time that an impact continues |
| **Phototrophic** | Obtaining most or all of an organism's energy requirements from sunlight via photosynthesis, usually from intra cellular microalgae. |
| **Possible effects guideline** | Pressure intensity that might cause effects on benthic biota. Equates to the likely worst-case (EPO). |
| **Prediction** | A forecast of future outcomes |

| Word, phrase or acronym | Definition for the purpose of this Technical Guidance |
|---|---|
| **Pressure threshold** | Pressure thresholds signify a level of pressure (generally expressed in terms of intensity, frequency and duration) that equates to a pre-defined level of effect or impact to an organism or group of organisms of interest. |
| **Probable effects guideline** | Pressure intensity that is likely to cause effects on benthic biota. Equates to the likely best-case (MT). |
| **Recoverable** | See reversible |
| **Reversible** | A capacity to recover or return to a state resembling that prior to being impacted within a timeframe of five years or less. |
| **ROV** | Remotely operated vehicle |
| **Serious damage** | Damage to benthic communities and/or their habitats that is effectively irreversible or where any recovery, if possible, would be unlikely to occur for at least five years. |
| **Severity** | The degree of harm caused. For example, the degree of harm or severity of impact to biota could range from sublethal effects to mortality or loss. |
| **SSC** | Suspended sediment concentration |
| **State coastal waters** | The State coastal waters extend three nautical miles seaward from the territorial sea baseline. |
| **Uncertainty** | In relation to prediction is doubt or concern about the reliability of achieving predicted outcomes. |
| **WAMSI** | Western Australian Marine Science Institution |
| **Worst-case** | See Likely worst-case |
| **ZOHI** | Zone of High Impact. The areas where serious damage to benthic communities is predicted or where impacts are considered to be irreversible. |
| **ZOI** | Zone of Influence. The areas where changes in environmental quality associated with dredge plumes are anticipated during the dredging operations, but where these changes would not result in a detectible impact on benthic biota. |
| **ZOMI** | Zone of Moderate Impact. The areas where predicted impacts on benthic organisms are sub-lethal, and/or the impacts are recoverable within a period of five years following completion of the dredging activities. |

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# Appendix A: Guidelines to predict and manage the impacts of dredging

## Contents

# 1 Introduction

Numerical guidelines that signify critical levels of dredging pressure to key species groups, based on biological response thresholds, are presented in this appendix. These thresholds are typically based on scientific knowledge derived through experimentation in the laboratory under simulated dredging conditions, and from field measurements/observations taken under actual dredging conditions. The Dredging Science Node of the Western Australian Marine Science Institution (WAMSI DSN) research program provides insight to help derive guidelines that are applicable to the Western Australian marine environment. The relevant knowledge and suggested 'possible-effects' and 'probable-effects' guideline values (see section 3.5 of the main body of this technical guidance for explanation of these terms) for corals, seagrasses and sponges are presented below.

It is important to recognise that these values are provided as a guide only. The final values chosen are the responsibility of the proponent and will be influenced by the nature and scale of the proposed activities, the regional location, resident BCH and natural background conditions.

# 2 Corals

The key pressure-response pathways linking dredging to coral health are strongly influenced by coral morphology (see Jones et al 2019a for overview of findings of DSN research on corals). The critical indicators of dredging pressure on corals were found to be light limitation caused by the shading effects of the sediment suspended through the water column and sediment deposition on coral surfaces as those sediments settle out. The main pathway by which dredging impacts erect and branching coral morphologies, such as *Acropora* spp., is via reduced light availability caused by suspended sediments in the water column. Deposited sediment can also add stress to these species, but this is less important as their morphologies are not conducive to retention of sediment. Further, the concentrations of suspended sediment required to have a significant deposition effect on this morphological type during a dredging program are such that the light attenuating properties would impart severe light limitation (see Fisher et al 2017a).

In contrast, species with more foliose growth habits (eg. *Turbinaria* spp.) and the massive dome-shaped taxa such as *Porities* spp. are likely to be more affected by sediment that has fallen out of suspension and deposited on the surface of the colony. These morphologies are more conducive to retaining sediment, inhibiting solute exchange and reducing light reaching the endosymbiotic zooxanthellae. Deposited sediments can also cause necrotic lesions when they cannot be readily removed, either actively by muco-ciliary transport or passively by waves and currents. Active removal by the coral polyps comes with an inherent energetic cost, and once lipid energy reserves are depleted sediments can only be removed by waves and currents. These species are also affected by light limitation due to suspended sediment plumes, so the two pathways operate simultaneously.

Considered in isolation of other pressures, there is no evidence that adult corals are directly affected by sediments suspended in the water column through processes such as physical abrasion or clogging of the heterotrophic feeding apparatus, however suspended sediments do appear to produce some additional negative affects when light levels are low and limiting. Suspended sediments do have measurable effects on some, but not all, coral reproductive processes (see section 3.7).

## 2.1 Outer boundary of ZOMI

### Considering the effects of light reduction on corals

WAMSI DSN experimental studies found a range in the low light tolerances of the different shallow water coral species tested, but was able to determine with high confidence that no impacts are expected to any corals if the daily light integral (DLI) is >4 mol quanta $m^{-2}$ $d^{-1}$ (Jones et al 2019b).

The guideline values in Table A1 are based on light levels and durations of exposure to those levels that caused 10% dis-colouration (~ bleaching) in corals under controlled laboratory conditions (EC10). The 20- and 30-day 'possible-effects' guidelines reflect the lowest pressure that could be reasonably expected to cause 10% bleaching for the most sensitive species (*Pocillopora acuta*) of the two species tested (Bessel-Browne 2017). The 5-day 'possible-effects' guideline is based on an observation of whitening occurring after 4-5 days in darkness. The 'possible-effects' guideline values represent thresholds where impacts are possible (but un-likely) and are recommended for use to define the worst-case boundary for corals when light reduction is the only consideration. The 'probable-effects' guidelines are identical except they are based on the species that is most tolerant of reduced light availability (*Acropora millepora*) of the two species tested (Bessel-Browne 2017).

The guidelines have three temporal components that must all be met simultaneously. For example, this means that for the possible effects guidelines not to be triggered (i.e. no effects): during any running 5 day period the average DLI should be 0.1 or greater and for any running 20 and 30 day periods the average DLI should be 2.3 and 2.8 or greater respectively.

**Table A1: Possible-effects and probable-effects guideline values for the ZOMI for corals when the only consideration is light reduction. Adapted from Bessel-Browne et al (2017) and applying professional judgment.**

| Guideline type | Daily Light Integral (mol photons $m^{-2}$ $d^{-1}$) | Duration (days) |
|---|---|---|
| Possible effects | <0.1 | >5 |
| Possible effects | <2.3 | >20 |
| Possible effects | <2.8 | >30 |
| Probable effects | <0.1 | >10 |
| Probable effects | <1.6 | >20 |
| Probable effects | <1.9 | >30 |

### Predicting the combined effects of light and SSC (and deposited sediment) on massive and foliose corals

Under controlled conditions where corals were subject to varying combinations of DLI and SSC, but sediment deposition was prevented, the most sensitive appeared to be the branching species (*Pocillopora damicornis* > *Acropora millepora*) followed by the massive corals (*Porites spp.*). The foliose coral *Turbinaria reniformis* is often found in relatively turbid environments and was the most tolerant to elevated SSC of all species tested. Overall, the experimental results suggest that if SSC concentrations remain below 10 mg $L^{-1}$ and if daily light levels are maintained at >2.2 DLI there may be some sub-lethal effects but no coral mortality would be expected (Jones et al 2019b). In considering these values it is important to note they were derived under controlled conditions where sediment accumulation was prevented.

In contrast to the experimentally-derived thresholds described above, the guideline values in Table A2 implicitly account for sediment deposition effects because they were derived from analyses of *in situ* water quality and coral health data associated with a large-scale capital dredging project conducted in a relatively clear offshore location in the Pilbara (Barrow Island). As such they are considered to include the additive effects of sediment deposition, elevated SSC and reduced light availability (Fisher et al 2019).

The possible-effects guidelines represent a set of light and turbidity/SSC levels that if triggered would possibly result in some level of coral mortality. To determine whether the possible effects guideline values have been triggered, the average DLI and NTU or SSC measurements need to be compared to the triggers for each of the averaging periods (3, 7, 10, 14 and 28 day averaging periods). So using the 3 day averaging period for an example, average DLI should be ≥1.1 mol photons $m^{-2}d^{-1}$ and NTU or SSC (whichever is measured) should be ≤10.8 units or ≤19.4 mg $L^{-1}$ respectively to ensure no coral mortality. Similarly, the probable-effects guidelines represent levels that if triggered will probably result in some level of coral mortality.

**Table A2: In-situ derived possible-effects and probable-effects guideline values for the ZOMI for corals when considering light reduction and turbidity/suspended sediments in combination. Adapted from Fisher et al (2019). Suspended sediment concentration (SSC) units are mg $L^{-1}$ and daily light integral (DLI) units are mol photons $m^{-2}$ $d^{-1}$. NTU is nephelometric turbidity units. SSC was calculated from NTU where SSC= NTU x 1.8.**

| Threshold type | Averaging period | Possible-effects | | |
|---|---|---|---|---|
| | | NTU | SSC | DLI |
| Running mean (days) | 3 d | >10.8 | >19.4 | <1.1 |
| | 7 d | >8.2 | >14.7 | <1.8 |
| | 10 d | >7.3 | >13.1 | <2.2 |
| | 14 d | >6.5 | >11.7 | <2.5 |
| | 28 d | >5.2 | >9.3 | <3.1 |

| Threshold type | Averaging period | Probable-effects | | |
|---|---|---|---|---|
| | | NTU | SSC | DLI |
| Running mean (days) | 3 d | >19.9 | >35.7 | <0.3 |
| | 7 d | >13.6 | >24.5 | <0.6 |
| | 10 d | >11.6 | >20.9 | <0.9 |
| | 14 d | >10.0 | >18.0 | <1.1 |
| | 28 d | >7.3 | >13.2 | <1.8 |

It should be noted that the SSC triggers in Table A2 were calculated from measured turbidity (in NTU) by applying a site-specific conversion factor where SSC = NTU x 1.8. Typically, SSC concentrations will be predicted from sediment generation and fate modeling whereas turbidity is an optical measurement of water clarity typically measured in the field using sensors and reported as NTU. Marine water clarity is controlled primarily by coloured dissolved organic matter (CDOM), phytoplankton and sediment particles suspended in the water column. There are typically low levels of CDOM and phytoplankton in Western Australian coastal waters, and as such turbidity is primarily influenced by SSC, particularly where sediments are actively released to the water column through dredging-related activities.

When applying the guideline values in Table A2 for dredging impact prediction purposes it should be noted that although these parameters are related, the key variable is DLI which, for a given level of turbidity, is depth dependent. Therefore, meeting the guidelines in Table A2 for SSC/turbidity does not necessarily mean that the corresponding DLI guidelines will be met. For EIA purposes, appropriate resolution bathymetry for areas of BCH, and the site-specific algorithms to describe the relationship between light attenuation coefficient (LAC) and SSC, will be needed to calculate benthic DLI for interrogation against the guidelines. Similarly, the relationship between LAC and turbidity will be needed to calculate suitable turbidity triggers for subsequent monitoring and management.

There are few relevant published data on relationships between TSS and LAC in Western Australia. Fearns et al (2019) examined the relationship between TSS and extinction coefficient[10] (Kd) during a large-scale dredging campaign in the mid-shore Pilbara region near Onslow. They found significant differences in attenuation of different wavelengths within the PAR spectral band (400 - 700 nm) and that the extinction coefficient ($m^{-1}$) at a wavelength of 490nm could be derived from TSS concentration (≥ 3mg/L) according to the following equation:

$K_{d490} = 1.018 (\ln(TSS)) - 0.865$

The guidelines in Table A2 are particularly suited for use in clear-water environments (e.g. offshore Pilbara). Even in these environments it will be necessary to ensure that the relationship(s) between sediments suspended by dredging and turbidity are established for the local area. Caution should therefore be used if applying the guideline values in Table A2 to naturally turbid waters that typify many Pilbara in-shore environments. It is probable that the guideline values are naturally exceeded in these areas at times, and where corals occur in these areas the dominant species are likely to be the more turbidity-tolerant massive and foliose forms (e.g. *Turbinaria* spp.). If this is the case then any guidelines proposed for predicting impacts in these areas should take these issues into account and be tailored accordingly. Advice on developing generic guidelines using background water quality data is provided in EPA (2016b). Advice on developing guidelines for corals from background water quality data is provided in Fisher et al. (2019) and is discussed in section 4.2.1 of Appendix C in the context of dredging management.

## 2.2 Outer boundary of ZOHI

### Considering the effects of suspended and deposited sediment, and light availability

In areas close to dredging and disposal activities, sediment deposition becomes an important potential impact pathway. Experiments show that most coral species and morphologies tested were capable under slight water-flow (<3 cm $s^{-1}$) of removing all sediment up to 20 mg $cm^{-2}$ $d^{-1}$ leaving only slight residual deposits typically less than a few percent of the surface area. The circular massive morphology (e.g. *Goniastrea retiformis*) could do the same up to 40 mg $cm^{-2}$ $d^{-1}$. The branching species *A. millepora* managed to clear a sedimentation rate (under static conditions) of up to 235 mg $cm^{-2}$ $d^{-1}$, an order of magnitude higher than the other morphologies.

Sediment deposition is difficult to predict and measure in ecologically relevant terms but DSN research has identified that in dredging situations it is closely coupled to SSC (see Fisher et al 2019). The high SSC required to cause high sediment deposition rates will also cause significant light attenuation, and reduced light availability at the seabed. Corals have been shown to bleach after 10 days under very low light, and fully bleach after 20 days. When corals maintained under zero or very low light are simultaneously exposed to high total suspended sediment concentrations (e.g. SSC of 100 mg $L^{-1}$), the effects can be greater than when exposed to low light alone (Bessell-Browne et al 2017b). The causal mechanism for this ‘additive’ effect is currently unknown.

[10] Extinction coefficient (natural log) can be converted to LAC (log10) by applying a factor of 0.435.

Bleaching reduces the amount of energy provided by the zooxanthellae so they need to access stored lipids to survive. Active sediment clearance comes at a metabolic cost so it is unlikely corals could withstand significant levels of sediment deposition for very long once bleached and their stored energy reserves have been consumed.

The SD and DLI guidelines in Table A3 are derived from the considerations outlined above. The NTU values have been extrapolated from the relationship between NTU and DLI generated by combining the data in Table A2. SSC was calculated from NTU where SSC = NTU x 1.8. The SSC values are provided to assist in impact prediction and the NTU values for monitoring purposes. The SSC guideline of 70 $mgL^{-1}$ in Table A3 would equate to an extinction coefficient of 1.47 $m^{-1}$ (based on the relationship in Fearns et al 2019) for the 490 nm wavelength. Without considering the additional attenuation associated with phytoplankton and water colour, 490 nm light would be rapidly attenuated under these conditions and if incident light was 1000 umol photons $m^{-2}$ $s^{-1}$ at the water surface[11], there would be zero photosynthetically-usable light at a depth of ~ 2.04 m.

**Table A3: Possible-effects and probable-effects guideline values for the ZOHI for corals when considering sediment deposition, light reduction and turbidity/suspended sediments in combination. Sediment deposition (SD) is mg $cm^{-2}$ $d^{-1}$, suspended sediment concentration (SSC) units are mg $L^{-1}$ and daily light integral (DLI) units are mol photons $m^{-2}$ $d^{-1}$. NTU is nephelometric turbidity units.**

| Threshold type | Averaging period | Possible-effects | | | |
|---|---|---|---|---|---|
| | | SD | NTU | SSC | DLI |
| Running mean (days) | 10 d | >20 | >38 | <70 | <0.1 |

| Threshold type | Averaging period | Probable-effects | | | |
|---|---|---|---|---|---|
| | | SD | NTU | SSC | DLI |
| Running mean (days) | 20 d | >40 | >38 | <70 | <0.1 |

Caution should be exercised when applying these guidelines as the ‘effects’ being considered are significant levels of mortality/serious damage. Furthermore, conditions that do not trigger the guidelines set out above may still cause serious impacts including mortality and damage. A very conservative and far more risk averse approach for delineating the ZOHI would be to apply the probable-effects guidelines in Table A2 that signify some level of coral mortality (or derive guidelines from these using professional judgement e.g. by applying appropriate multiplication factors). In reality, the most appropriate guidelines probably lie somewhere in between the values in Tables A2 and A3, but there is currently insufficient understanding available to confidently determine a threshold amount and duration of dredging-related pressure that, if exceeded, will cause serious and/or irreversible damage to corals.

Proponents are reminded that the ZOHI is defined in terms of recovery potential over a specified timeframe and they will need to consider this when deciding how to predict the ZOHI and any guidelines they may use to assist in making the prediction. Furthermore, it is expected that more information will be gathered during future dredging programs and made available via IMSA to assist in these determinations.

[11] for context - full sunlight intensity at midday is typically ~2000 umol photon $m^{-2}$ $s^{-1}$

## Predicting recovery of impacted coral communities

As discussed earlier in sections 3.2 and 3.4, the ZOHI is defined by whether the impacted benthic communities are predicted to recover within a specified timeframe. DSN research used a combination of *in-situ* demographic measurements and scenario modeling to better understand recovery mechanisms in coral communities and provide an indication of the levels of mortality of existing corals and levels of recruitment failure that could be tolerated and still provide some likelihood of recovery within five years post disturbance (Babcock et al 2017).

The research concluded that it should not be assumed that recovery from impacts to adult colonies will be rapid, and in the case of shorter-lived species such as *Acropora* spp., impacts to recruitment levels appear to be particularly important and have significant additional consequences for recovery. Small impacts on recruitment processes, if sustained over long periods, will have greater long-term impacts on population density and cover than single severe events. *Porities* colonies can be very long-lived, and particular effort should be placed on identifying and avoiding/minimizing damage to large *Porities* colonies, as recovery of *Porities* populations, when considered in terms of size structure and ecological role in providing structure on reefs, could take a very long time (e.g. decades to centuries).

The guideline values in Table A4 are derived from two years of sampling at a single location in the Pilbara and the application of professional judgment, and as such they are best considered as Low Reliability Guidelines[12]. Furthermore, the level of confidence that can be attributed to predictions of recruitment failure for EIA purposes is unclear. Nonetheless Table A4 has been included here to provide some guidance for proponents to use for predictive purposes, and for interpreting monitoring results during project execution for compliance purposes, in the absence of site-specific information on demographic processes occurring in a particular location.

**Table A4: Possible-effects and probable-effects guidelines for the ZOHI for corals when considering the role of demographic processes in the recovery of coral communities from disturbance within five years and in the absence of location-specific knowledge. Triggering the guidelines at a location (e.g. ≤10% mortality or ≤ 20% reduction in recruitment in any year for A. millepora) would signify it is in a ZOMI; exceeding the guidelines at a location would signify it is in a ZOHI. Adapted from Babcock et al (2017) and applying professional judgment. Percentages are relative cover.**

| | | Species | | |
|---|---|---|---|---|
| | | ***Acropora millepora*** | ***Turbinaria mesenterina*** | ***Porities* spp.** |
| Guideline type | Pressure | | | |
| Possible effects | Uniform mortality* – single event | ≤10% | ≤10% | ≤5% |
| Probable effects | Uniform mortality* – single event | ≤15% | ≤15% | ≤7.5% |
| Possible effects | Recruitment reduction^ (% and # of years) | ≤20% 1yr | ≤80% 1yr | ≤95% 1yr |
| Probable effects | Recruitment reduction (% and # of years) | ≤50% 1yr<br>≤20% ≤2yr | ≤80% ≤3yr | ≤95% ≤3yr |

* applying uniformly to all size classes

^ This means % reduction in *recruitment, not % recruitment*

[12] see ANZWQG (2018) for a description of this term as it relates to the derivation of water quality guidelines.

# 3 Seagrasses

Deterioration in light availability caused by suspended sediments is the main pressure-response pathway by which dredging impacts seagrasses, and the effects can extend considerable distances from excavation and disposal sites (see Lavery et al 2019 for overview of findings of DSN research on seagrasses). Tropical seagrasses require sufficient light to maintain a positive energy balance over timeframes of weeks to survive. However, even when levels are sufficient on average, seagrasses cannot tolerate a number of consecutive days of little or no light. Burial or covering of seagrasses by sediments containing moderate levels of organic matter (≥ 4% OM) can affect seagrasses directly when deposition rates are very high, but this is only likely to occur in areas directly adjacent to dredging or disposal sites.

Altered spectral quality of light, such as caused by sediments suspended during dredging, can also affect seagrass vigour and seed germination (sometimes positively), but the shift in PAR spectral quality is considered a secondary issue in comparison to reductions in PAR quantity. There is no evidence that suspended sediment generated by dredging affects seagrass directly through mechanisms such as physical abrasion.

## 3.1 Outer boundary of ZOMI

### Considering the effects of light reduction on seagrasses

Light reduction threshold values for lethal effects were derived for three seagrasses *Cymodocea serrulata, Halodule uninervis* and *Halophila ovalis* that commonly co-occur in NW WA using the empirically measured variables of total and above-ground biomass. These were found to be the most appropriate lethal bioindicators based on a variety of approaches trialed during two experiments.

The first experiment determined the responses of seagrasses to a gradient in light availability, spanning the range predicted to occur close to dredging operations, as well as light levels expected to elicit a mortality response determined from published research. The second experiment confirmed that, in addition to the ‘average’ level of light availability, the ‘pattern’ of light delivery affects how *C. serrulata* and *H. uninervis* respond to a reduction in light availability and their capacity to recover.

Recovery potential for a given level of ‘average’ reduced light availability was greater when the pattern of delivery included frequent intervening periods of moderate/high light compared to patterns providing short respite periods of high light. Therefore, designing dredging programs to minimise the number of consecutive days of low light would likely result in lower impacts than dredging programs that cause long periods of low light.

The guideline values in Tables A5 and A6 are equivalent to no observable effects concentrations (NOEC) for light-mediated impacts due to suspended sediments in the water column, but with higher and lower levels of confidence respectively. They can be used to help predict the boundary between the ZOMI and the ZOI. The possible-effects guidelines in Table A5 suggested for describing the EPO boundary are thresholds where there is high confidence that seagrass will be protected from impact; the probable-effects guidelines in Table A6 suggested for describing the MT boundary represent thresholds where there is lower confidence that seagrass will be protected from impact.

The guidelines have two components: 1) an average light intensity over a two-week averaging period with durations ranging from three to 12 weeks, and 2) a maximum permissible period of low light during each two-week averaging period. If minimum daily average light availability does not trigger the relevant guideline value during every two-week period for the duration (e.g. for 12 weeks), **and** the maximum periods of low light within each and every two-week averaging period does not trigger the guideline values, it is unlikely that seagrasses would be measurably affected.

**Table A5: Possible-effects guideline values to define the outer boundary of the ZOMI for *Cymodocea serrulata, Halodule uninervis, Halophila ovalis* and mixed seagrass assemblages for use as an EPO. Daily light integral (DLI) units are mol photons $m^{-2}$ $d^{-1}$. See footnote and Lavery et al (2019) for explanation of the derivation of the guideline values.**

| | Two-week averaging period over the specified duration | | Within a two-week averaging period | |
|---|---|---|---|---|
| | Duration (weeks) | Mean DLI (for each two-week period) | Duration (days) | Mean DLI (daily) |
| ***Cymodocea serrulata*** (based on **above-ground biomass**) | | | | |
| Possible effects | >12 | <8.9 | 5 | <2 |
| Possible effects | >9 | <2.3 | 5 | <2 |
| ***Halodule uninervis*** (based on **above-ground biomass**) | | | | |
| Possible effects | >12 | <13.1 | 5 | <2 |
| **Mixed Meadow** (based on **total biomass** of all species in a multi-species meadow) | | | | |
| Possible effects | >12 | <13.1 | 5 d | <2 |
| ***Halophila ovalis1*** | | | | |
| (based on **above-ground** biomass) | | | | |
| Possible effects (interim) | >3 | <0.9 | | |

[1] The experimental results for *H. ovalis* were inconclusive and thresholds could not be developed from the data. Instead, an interim threshold was developed following the guideline recommendations for biological indicators (ANZWQG 2018), where impact conditions were compared to background or reference conditions. This approach simply indicates the amount of light reduction that would cause the median value for that variable at an 'impact' site to fall below the 20th percentile for that variable at a valid control site.

**Table A6: Probable-effects guideline values corresponding to the outer boundary of the ZOMI for *Cymodocea serrulata, Halodule uninervis* and mixed seagrass assemblages for use as a MT. These guidelines can be considered to be 'no observable effects levels' but with lower confidence than the possible-effects guidelines in Table A5. Daily light integral (DLI) units are mol photons $m^{-2}$ $d^{-1}$. The guidelines were experimentally determined (see Lavery et al 2018).**

| | Two-week averaging period over the specified duration | | Within a two-week averaging period | |
|---|---|---|---|---|
| | **Duration (weeks)** | **Mean DLI (for each two-week period)** | **Duration (days)** | **Mean DLI (daily)** |
| ***Cymodocea serrulata* and *Halodule uninervis* (based on above-ground biomass)** | >12 | <2.3 | >10 | <2 |
| **Mixed Meadow** (based on **total biomass** of all species in a multi-species meadow) | >12 | <8.9 | >10 | <2 |
| | >6 | <5.0 | >10 | <2 |

### Considering the effects of burial on seagrasses

The guideline values in Table A7 represent no observable effects levels of organic rich sediment deposition that can be tolerated by two seagrass species under low light conditions and would be suitable for defining an EPO boundary. These conditions of low light and high rates of sediment deposition are likely to be restricted to the close vicinity of dredging and disposal areas but the lack of reliable field data makes it difficult to determine where, and how often, these conditions occur. It should be noted that these species did not show any negative responses to burial by inorganic sediments under the same conditions. The organic content of capital dredging material (e.g. cut limestone) is likely to be very low compared to unconsolidated surface sediments that are more typical of maintenance dredging activities.

**Table A7: No observable effects sediment burial thresholds for *Cymodocea serrulata* and *Halodule uninervis*. The values are based on the depths and durations of burial in organic rich sediments (4% O.M.) that can be tolerated under low light conditions (2.45 mol photons $m^{-2}$ $d^{-1}$). These are considered possible-effects guideline values that can be used to determine the outer boundary of the ZOMI for use as an EPO.**

| Maximum depth of burial | Maximum period of burial | Sediment type (% organic matter)* | Ambient light (mol photons $m^{-2}$ $d^{-1}$) |
|---|---|---|---|
| ≤40 mm | ≤6 weeks | ≤4% OM | ≥2.45 |

*Neither species showed negative responses to burial by inorganic sediments under the same conditions.

## 3.2 Outer boundary of ZOHI

Tropical seagrass plants are typically short-lived annuals or perennials, and hence the boundary between the ZOMI and the ZOHI is effectively determined on the basis of predictions of whether or not an impacted meadow will recover within five years after dredging ceases, rather than on thresholds of pressure that generate mortality.

When conditions are favourable, most tropical seagrasses have the ability to colonise suitable habitat, grow and spread rapidly. It is reasonable that impact assessments incorporate the possibility that meadows can be severely impacted in the short term (i.e. effectively all vegetative material lost) but recover in the longer term through re-growth from remnants, immigration or from seed banks.

Persistent and viable seed banks have been recorded over extensive areas along the Dampier Peninsula in the west Kimberley, and provide for the annual re-establishment of new meadows after the wet season die-off. As such there is a strong body of evidence to support the assumption that these meadows could also recover if impacted by dredging activities.

DSN research focussed on the lesser-known seagrasses of the Pilbara and identified that for most meadows, both sexual reproduction and vegetative growth are important for maintaining populations; and there was a reasonably high level of migration of genes over distances of 2–5 km, but lower levels over greater distances. However the research has shown that, in contrast to the west Kimberley, predictions about recovery of seagrass in the Pilbara should not assume a priori that there will be a rapid recovery from seed banks. While one cannot discount that seagrass populations could recover rapidly following disturbance, there is no evidence in the Pilbara of extensive and persistent seed banks that would provide a mechanism for that recovery. If field sampling demonstrates the presence of viable seed banks then there would be greater confidence that recovery via seed was possible and probable.

A primary mechanism for recovery after disturbance is through extension of vegetative growth from surrounding meadow(s) into disturbed patches. Therefore, seagrass losses that encompass relatively small areas, typical of that due to anchoring of vessels and deployment of equipment, is likely to be temporary in meadows comprised of colonising species, such as *Halophila* spp., as significant regrowth could be expected within a year and it would be reasonable to assume full recovery within five years.

While it is plausible that recovery through rhizome extension could be achieved for un-vegetated patches within meadows, this is less likely for losses that encompass hectares. Similarly, recovery is possible from seagrass fragments that have been dislodged elsewhere and drifted into the impacted area, however the likelihood of this occurring is considered low and from an EIA perspective should not be relied upon as a recovery mechanism.

Based on the current evidence, if areas of seagrass in the Pilbara are predicted to be lost, and there is no evidence of persistent and viable seed banks, then those areas would be deemed to be in the ZOHI. If areas of seagrass are predicted to be severely impacted, but viable seed banks have been shown to be present or some remnant but viable vegetative material is expected to remain within those areas after the cessation of dredging, it would be appropriate to designate those areas as ZOMI (see Table A8).

**Table A8: Using predicted impact and recovery potentials to differentiate between the ZOHI and the ZOMI for seagrasses. These values could be considered possible-effects guidelines and used to determine the outer boundary of the relevant zone for use as an EPO.**

| Level of predicted impact | ZOHI | ZOMI |
|---|---|---|
| Total loss | No viable seed bank present | Viable seed bank present |
| Area of partial loss (i.e. live seagrass adjacent to loss area and where no viable seedbank present) | ≥ 0.25 ha | < 0.25 ha |

# 4 Sponges

Sponges are a very diverse group of organisms, with a range of morphologies and nutritional modes and WAMSI DSN research has not identified a clearly dominant dredging-related pressure-response pathway (see Abdul Wahab et al 2019 for overview of findings of DSN research on sponges). Sediments generated and released during dredging can have direct effects through sediment clogging the aquiferous systems in both photoautotrophic and heterotrophic species. The increased light attenuation from suspended sediments can reduce benthic light and associated photosynthesis in phototrophic sponges, but has no effect on heterotrophic species. Deposited sediment can smother sponges and adversely affect taxa that utilise either nutritional mode, by preventing feeding/ oxygen exchange and/or reducing photosynthesis.

The diversity of species and responses to dredging-related pressures, makes it difficult to predict the community level consequences of dredging activity. Sponges exhibit variable sensitivities to simulated dredging pressures (i.e. light attenuation, elevated SSC and smothering from deposited sediments) when considered singly and in combination under controlled conditions. There was a faster and greater level of response to the combined stressors (which is more realistic of conditions encountered during dredging) than when the stressors were applied individually.

Nutritional mode appeared to be the most important factor to consider when evaluating the likelihood (and extent) of dredging related impacts, with phototrophic sponges being most sensitive. Across all dredging pressures, the phototrophic cup sponge *Carteriospongia foliascens* was the most sensitive species, exhibiting rapid bleaching and mortality under low light and high SSC conditions (Wahab et al 2017a). When impacted, phototrophic sponges are also less likely to recover after conditions improve. Some phototrophic species bleach within 72 h when held in complete darkness and suffer complete mortality even when provided with natural optimal light conditions post impact. The health of others is seriously impaired after 7–14 d in complete darkness but fully recover after 14 days of natural conditions. Some phototrophic species are more resilient and can fully recover from 28 days of complete darkness once returned to natural light conditions.

Morphology was also relevant, particularly under high sediment deposition conditions, with wide cup-shaped, massive and encrusting morphologies generally more susceptible than other morphologies that were less likely to accumulate sediment.

Overall, both phototrophic and heterotrophic sponge species can survive under moderately low light intensities (DLI ≤3.1 mol photons m-2 d-1) for periods of at least 28 days.

A detailed summary of how sponges respond to dredging-related stressors (under realistic conditions for dredging in NW WA), including experimentally derived stress thresholds are provided in Abdul Wahab et al (2019).

## 4.1 Outer boundary of ZOMI

### Considering the effects of light reduction on phototrophic sponges

The possible-effects and probable-effects guideline values in Table A9 can be considered 'no observable effects' levels and can be used to define the outer boundary of the ZOMI for EPO and MT purposes respectively when light reduction is the only consideration (relevant to deeper water and/or where SSC is consistently well below 10 mg/L and measurable levels of dredging-related sediment deposition are unlikely). The guidelines have three temporal components (short, medium and long) that must all be met simultaneously. For example, this means that to ensure the possible effects guidelines are not triggered, and phototrophic sponges are protected from impact (i.e. in a ZOI): the DLI should never fall below 0.1 for any non-consecutive 24 hour period and for any running 2–7 day period the average DLI should never fall below 1 and, during any running 8–28 day period the average DLI should be 3.1 or greater.

**Table A9: Possible-effects and probable-effects guideline values for phototrophic sponges to define the boundary between the ZOMI and ZOI for use as an EPO and a MT respectively and when considering the effects of light in isolation of other dredging related pressures. Note that the long-, medium- and short-term guideline values must all be met simultaneously. Daily light integral (DLI) units are mol photons $m^{-2}$ $d^{-1}$.**

| | Short term | | Medium term | | Long term | |
|---|---|---|---|---|---|---|
| Guideline type | Mean DLI | duration (days) | Mean DLI | duration (days) | Mean DLI | duration (days) |
| Possible-effects range | <0.1 | 1 | ≤1 | 2–7 | ≤3.1 | 8–28 |
| Probable-effects range | n/a | n/a | <0.1 | 2–7* | <1 | 8–28 |

* if *Carteriospongia foliascens* is present (or other sensitive phototrophs) then a duration of ≤2 days may be more appropriate.

### Considering the combined effects of light, suspended and deposited sediment on sponges

The guideline values in Table A10 represent no observable effects levels and can be used to define EPO and MT boundaries respectively when the effects of suspended sediment concentration, light reduction and sediment deposition are considered simultaneously. If none of the guidelines are triggered then it is likely that all sponge taxa will be protected (i.e. in a ZOI), however the light criteria can be omitted if there is confidence that phototrophic species are absent or a minor component of the assemblage.

**Table A10: Possible-effects and probable-effects guideline values to define the outer boundary of the ZOMI for all sponges for use as an EPO and a MT respectively and considering the effects of suspended sediment concentration (SSC) and sediment deposition as a single deposition event and as pulsed deposition events. Note that when phototrophic sponges are part of the assemblage all guideline values, including those in Table A9, must be met for the EPO/MT to be achieved.**

| Guideline type | SSC (mg $L^{-1}$) | Maximum single deposition (mg $cm^{-2}$) | Maximum pulsed deposition (mg $cm^{-2}$) | Pulsed deposition frequency |
|---|---|---|---|---|
| Possible-effects | >10 | >8 | >2 | >1 in 4 days |
| Probable-effects | >10 | >16 | >3 | >1 in 4 days |

## 4.2 Outer boundary of ZOHI

### Considering the combined effects of light, suspended and deposited sediment on sponges

The guideline values in Table A11 represent lowest observable (sub-lethal) effects levels, but are generally less conservative than the values in Table A10 and can be used to define the outer boundary of the ZOHI.

**Table A11: Possible-effects and probable-effects guideline values to define the outer boundary of the ZOHI for all sponges for use as an EPO and a MT respectively and considering the effects of suspended sediment concentration (SSC), light as a daily light integral (DLI) and sediment deposition as a single deposition event and as pulsed deposition events. Note none of the guideline values should be triggered for the EPO to be achieved when phototrophic sponges are part of the assemblage. The DLI guidelines can be omitted if phototrophic sponges are absent or in low abundance and no sensitive species are present.**

| Guideline type | SSC (mg $L^{-1}$) | DLI (mol photons $m^{-2}$ $d^{-1}$) | duration (days) | Maximum single deposition (mg $cm^{-2}$) | Maximum pulsed deposition (mg $cm^{-2}$) | Pulsed deposition frequency |
|---|---|---|---|---|---|---|
| Possible-effects * | >10 | <1 | 28 | >16 | >3 | >1 in 4 days |
| Probable-effects | >13.9 ** | <0.62 | 28 | >30 | >3 | >1 in 4 days |

* note: these are identical to the ZOMI/ZOI probable effects guideline values in Table A10

** note: this is based on the experimentally derived lower 95% C.I.$_{10}$ of the LC10 for the most sensitive species tested

# Appendix B: Windows of environmental sensitivity

## Contents

# 1 Background

Windows of environmental sensitivity are generally linked to critical life cycle processes such as reproduction and recruitment (see section 3.7). Most species have critical windows of sensitivity, but the specific details are not well known. The WAMSI DSN undertook research to identify and collate relevant information to better understand and manage the potential impacts of dredging on critical life cycle processes of important marine taxa in Western Australia (WA). The key findings of the research on corals, fishes and other taxa are presented below.

# 2 Corals

Most reef-building corals in north-west Western Australia (NW WA) participate in a predictable, annual, multi-specific and synchronous mass-spawning event during a neap tidal cycle, 6–10 days after the full moon in March (Autumn). There is some inter-annual variability and latitudinal variation in this pattern, and also some spawning during spring but to a lesser degree. Adult corals release buoyant egg and sperm bundles during the spawning event that dissociate after rising to the water surface. Fertilization and embryogenesis occur in the water column. The growing embryos and larvae remain planktonic until reaching the final planula stage, where they search for suitable benthic substrate to attach to and grow into adult corals. In brooding corals, fertilization occurs internally and developed larvae are released over a number of months.

A component of the DSN research program was designed to gain a better understanding of 1) coral spawning patterns in WA (see Gilmour et al 2017) and 2) the relative sensitivity (and pressure-response pathways) of each of the early life stages of coral to dredging-generated sediments (see Ricardo et al 2018). The objective was to provide a more objective basis for decision making by proponents and regulators. The following sections are based on the outputs of the research coupled with professional judgment.

## 2.1 Coral spawning patterns in Western Australia

General patterns in the timing of coral spawning in WA can largely be attributed to variation in: 1) community composition; 2) latitude; and, 3) the timing of the full moon (see Gilmour et al 2017 and Table B1).

The most obvious differences in reproductive patterns are due to the relative abundance of brooding versus spawning corals in the population. The timing of gametogenesis and planulation (larval release) **of brooding corals** (i.e. releasing larvae; e.g. *Isopora, Seriatopora* and *Stylophora*) is not well established but is thought to occur over several months within the year, most likely around the full moon during most months through spring to autumn.

The primary period of spawning **of broadcasting species** (i.e. releasing gametes; e.g. *Acropora, Montipora* and *Goniastrea*) on all WA reefs is in autumn, primarily during March and/or April. Some species participate in a second multi-specific spawning during spring (October and/or November) on some northern WA reefs. Biannual spawning during autumn and spring appears to be more prevalent at lower latitudes, with prevalence decreasing with increasing latitude. Spawning is primarily annual from Ningaloo Reef south.

Typically, the autumn spawning is split between March and April in almost all years, with the majority of spawning occurring seven to nine nights after the full moon in March and coinciding with neap, nocturnal, ebb tides. March is normally the month with the highest spawning activity, however in some years when the full moon is later in the month, spawning may be more evenly split between March and April or may occur predominantly in April.

**Table B1: General patterns of coral spawning in Western Australia. Derived from Gilmour et al (2017)**

| Coral taxa | Frequency | Timing | Moon Phase | Region |
|---|---|---|---|---|
| Brooding corals (e.g. *Isopora, Seriatopora* and *Stylophora* | Monthly | Spring to Autumn | 6-10 Nights after full moon | All WA Reefs |
| Broadcasting corals (e.g. *Acropora, Monitpora* and *Goniastrea*) | Biannual for northern reefs | Primary event in Autumn – March and April<br>Secondary Event in Spring – October and/or November | 7-9 Nights after full moon | North of (and including) Ningaloo Reef |
| | Annual for southern reefs | Autumn | 7-9 Nights after full moon | South of Ningaloo Reef |

### 2.1.1 Advice for determining reproductive patterns in coral communities

In areas where there is limited understanding of coral reproduction, sampling may be required to determine or confirm coral reproductive patterns in the area of interest, particularly for long-duration projects where proponents cannot readily avoid the broad windows set out in Table B1. Prior to designing a sampling regime, proponents should review the relevant information (from the scientific literature and identified through IMSA) on patterns of reproduction for the relevant region, and the associated methodologies used to determine those patterns, for background and context.

The following sections provide general advice on designing sampling programs to characterise coral communities and determining the significance of spawning periods. Further and more detailed information can be found in Gilmour et al (2017).

#### Characterising coral communities for spawning assessments

Pre-development surveys should be undertaken to initially quantify community composition so that reproductive assessments account for the relative cover of different coral groups. Data on abundance and reproduction can be combined to identify the significant periods of reproductive output within a year at the level of the entire community. Species diversity can be high, and a convenient cut-off point can be chosen according to their cumulative contribution to total coral cover (e.g. 80%); consideration must also be given to whether certain species, although low in relative abundance, play a critical role in ecosystem function.

Although identification to the finest taxonomic resolution is always desirable, it may not be practical except for species that are particularly abundant and are easily identifiable from photographs and in situ. For other species, a practical approach to quantifying composition and reproduction in coral communities may be to group species according to a higher taxonomic level (e.g. Genus, Family) and to also consider growth form (e.g. massive, branching, encrusting) and reproductive mode (spawner, brooder).

Sampling design

In most instances, and without a comprehensive understanding of the local situation, dominant corals should be sampled throughout the potential reproductive season(s) to construct a time-series that demonstrates the development of gametes and their subsequent disappearance after spawning.

A sampling program to determine the proportion of species and colonies spawning or releasing planulae throughout the year should span several months from the start of spring to the end of autumn (i.e. no sampling is required in winter). Oogeneic cycles in spawning corals take several months, so in species known to spawn biannually (March and October) or over a protracted period (September to April) eggs will be present in the population during most months.

It is important to note that the absence of eggs in a colony provides few insights into broader patterns of reproduction, and additionally not all polyps within a colony may be reproductive, so it is advisable to maintain adequate levels of replication (i.e. number of individual colonies) and to take multiple samples from single colonies.

Several years of relevant data may be required to obtain sufficient understanding of a particular locality to allow sampling effort to be reduced while retaining adequate confidence.

Determining significance of spawning periods

Estimates of reproductive output in different months of the year can be obtained by combining the relative abundance of coral groups with the proportion spawning or releasing larvae. Once the relative estimates of abundance and reproductive timing are obtained for common taxonomic groups, the data can be combined to produce estimates of reproductive output by the community throughout the year.

The assemblage of corals that best characterises the reef coral community should be primarily based on percentage cover data.

Rare corals can be excluded as they generally only comprise a small proportion of the total cover. This significantly reduces effort, as finding and sampling rare species with sufficient replication is most time consuming and could cause proportionally significant damage to the local population.

The decision rules for establishing the threshold for including dominant species in a 'community' and the means by which they are categorised should be well documented (see section above on characterising coral communities).

It would be beneficial if a publicly accessible data-base of spawning patterns on different reefs was established to collate the relevant information and improve understanding over time. Any such data-base should have good quality assurance procedures in place to allow appropriate inferences to be made.

## 2.2 Effects of sediments on coral recruitment

A number of experiments were conducted by researchers in the DSN to identify and quantify the key pressure-response pathways by which dredging could affect the early life-history stages (gametes, embryos and larvae) and life-cycle processes of corals from fertilization through to settlement (see Negri et al 2019 for over view of research). The effects of the three main pressures associated with dredging (i.e. elevated SSC, increased light attenuation and elevated sediment deposition) were examined.

The results showed that light attenuation had minimal effects on the life cycle processes examined and is not discussed further. Suspended sediments were found to have some effects on egg/sperm bundle ascent and fertilization, but not on coral embryo and larval development. Deposited sediments were found to affect larval settlement with the greatest effects on upward facing surfaces with crustose coraline algae (CCA).

Relevant advice for evaluating the potential effects of suspended and deposited sediments on coral reproduction and recruitment is provided below.

### 2.2.1 Advice for determining the effects of elevated SSC

Egg-sperm bundle ascent

Under conditions of elevated SSC, sediment particles could adhere to egg-sperm bundles, reducing their buoyancy and delaying or preventing ascent to the water surface where the bundle breaks down and eggs and sperm dissociate. Reductions or delays in ascent reduced the egg-sperm encounter rates and subsequently fertilisation. The effect was depth dependent for any given SSC as the number of times a rising bundle encounters a sediment particle increases with depth (Table B2).

**Table B2: Suspended sediment concentration thresholds of effect on the ascent of egg-sperm bundles and on egg-sperm encounter rates for *Montipora digitata*. Sediments were coarse-silt carbonates. (Adapted from Ricardo et al 2018).**

| | Effect | | | |
|---|---|---|---|---|
| | On egg/sperm bundle ascent | | On egg-sperm contact | |
| Depth (m) | $EC_{10}$ (SSC mg $L^{-1}$) | $EC_{50}$ (SSC mg $L^{-1}$) | $EC_{10}$ (SSC mg $L^{-1}$) | $EC_{50}$ (SSC mg $L^{-1}$) |
| 10 | 71 | 211 | 53 | 131 |
| 15 | 47 | 141 | 35 | 87 |

Although experiments found the ballasting effect increased with increasing sediment particle sizes (e.g. equivalent to coarse silt), the possibility that finer cohesive inshore sediments will preferentially bind and accumulate on rising bundles should not be discounted. The outputs of sediment transport models could be compared to the effects concentrations in Table B2 (or calculated for other depths; see Ricardo et al 2017a) to assess risk to successful bundle ascent and egg-sperm contact over the few hours associated with predicted spawning events.

Fertilization

Sediment did not adhere to eggs, but sperm could become entangled and stripped from the water column resulting in fewer egg sperm encounters and reduced fertilization rates. The effects are greater at lower sperm concentrations and for organic clay-rich sediments containing mucopolysaccharides. Pilbara inshore siliciclastic sediments had lower $EC_{10}$ values than carbonate sediments typical of more offshore areas (Table B3).

The outputs of sediment transport models could be compared to the effects concentrations in Table B3 to assess risk associated with fertilization (after successful bundle rise and egg-sperm dissociation), over the few hours associated with predicted spawning events.

**Table B3: Suspended sediment concentration thresholds of effect on fertilization success in *Acropora tenuis* at optimal ($10^5$ $L^{-1}$) and sub-optimal ($10^4$ $L^{-1}$) sperm concentrations for inshore siliciclastic and offshore carbonate sediments. (Adapted from Ricardo et al 2018).**

| | | Effect | | |
|---|---|---|---|---|
| | | Fertilization success | | |
| Sediment type | | Sperm Concentration ($L^{-1}$) | $EC^{10}$ (mg $L^{-1}$) | $EC_{50}$ (mg $L^{-1}$) |
| Inshore siliciclastic | Very fine silt | $10^4$ | 40 | 205 |
| | | $10^5$ | 80 | 414 |
| Offshore carbonate | | $10^4$ | 214 | >800 |
| | | $10^5$ | >820 | >820 |

The predicted effects of sediments on bundle ascent and subsequent egg-sperm contact could be combined to provide an overall assessment of potential effects on fertilization success. Under typical circumstances it might be expected that even the lowest of these effects concentrations (~ 35–40 mg $L^{-1}$) would be quite localized and restricted to areas in relatively close proximity to dredging operations.

### 2.2.2 Advice for determining the effects of deposited sediments

Deposited sediments directly and indirectly altered larval settlement characteristics. Larvae actively avoided sediment covered surfaces and sought sediment-free surfaces to settle on, such as vertical and downward-facing surfaces. Furthermore, even low to moderate levels of sediment deposition adversely affected the health of crustose coralline algae (CCA), which is an important inducer of coral settlement. This indirectly affected the settlement of coral larvae.

The issues (and residual uncertainties) outlined above make it difficult to provide definitive guidance and advice for EIA and management. Furthermore, the difficulties associated with measuring sediment deposition rates (and levels of deposited sediments) in the field, and associated problems with validating sediment transport and fate models (see Sun et al 2019), would complicate attempts to undertake scenario testing for impact prediction using experimentally-derived effects thresholds. Nonetheless, it is possible to provide some general indicative guidance to help assess and manage risk (Table B4 and described below).

From the experimental results (from grooved surfaces) it would be reasonable to assume that if deposited sediments were less than ~ 5 mg $cm^{-2}$ then direct and indirect effects would be negligible. Indirect effects (via effects on CCA) might be expected at levels >5 mg $cm^{-2}$ if sustained for a week or more. Deposited sediment levels of ≥30 mg $cm^{-2}$ would be likely to reduce larval settlement on the optimal upper surfaces of substrates by about 10%. Settlement on upper surfaces would be limited where levels of deposited sediment exceed 100 mg $cm^{-2}$, and unlikely to occur at levels above 180 mg $cm^{-2}$.

**Table B4: The likely effects of different levels of deposited sediment on the settlement patterns of *Acropora millepora* larvae on upper surfaces of benthic substrates. The values are based on professional judgment and considering the findings of Ricardo et al (2017b), but should be considered indicative only and not be equated to survivorship. Experiments were primarily conducted using either coarse siliciclastic or coarse carbonate silt. CCA is crustose coralline algae.**

| Deposited sediment (mg $cm^{-2}$) | Duration | Likely effect |
|---|---|---|
| ≤5 | episodic | negligible |
| >5 | for ≥1 week | reduced attractiveness for settlement due to bleaching of CCA |
| 5 – <30 | at time of settlement | minimal effects |
| 30 – <100 | at time of settlement | measurably reduced settlement on upper surfaces; increased settlement on sub-optimal surfaces |
| 100 – <180 | at time of settlement | Significantly reduced settlement on upper surfaces; previously suitable microhabitat (e.g. grooves) could fill with deposited sediments and become unsuitable |
| ≥ 180 | at time of settlement | No settlement on upper surfaces; settlement restricted to vertical and downward facing surfaces |

Pre-development surveys

Information collected during pre-development surveys can be used to assess risk and help formulate and refine management strategies to reduce threats from dredging activities on the key reproductive processes of egg-sperm bundle ascent, fertilization and coral recruitment.

The community composition and relative abundances of corals likely to be exposed to sediment plumes should be quantified to predict the likely timing of predicted spawning through appropriate literature and surveys (guidance is provided above), and to assess the relative significance of the spawning event(s).

The locations and proximity of coral reef habitats to the proposed excavation and dredge material placement sites, and the degree of topographic complexity within them, should be mapped and described. Uniform flat substrates are likely to be at greater risk than complex rugose substrates. Knowledge of coral habitats further afield and outside the influence of proposed activities can be used to provide a regional context for potential recruitment to assess risks and consequences of potential effects at local scales.

The climatology of current speeds and directions (needed for hydrodynamic and sediment transport/ fate modelling) should be interrogated for the likely spawning periods, and coupled with coral habitat distribution data, used to understand the likely transport pathways between potential gamete 'source' and larval 'sink' habitats.

Surveys should be undertaken to assess the *likely* characteristics of sediments released to the water column from dredging and disposal activities that gametes are likely to come into contact with as the SSC thresholds of effect are influenced by the characteristics of suspended sediments. Particular attention should be given to components of sediment that increase its cohesiveness, such as mineral clay, organic carbon and extracellular polymeric substances (EPS) content. These characteristics will be difficult to predict prior to dredging but there would be opportunities to collect data on the *actual* characteristics of suspended sediments by sampling the water column, or fines overflowing from hopper barges, at the site of dredging, and from within the hopper to characterise the sediments released at the dredge material placement sites prior to spawning events. The suspended sediment characteristics are likely to change as different substrates are dredged over time (inshore and offshore components of approach channels for example). This information would contribute to an evidence base to evaluate and/or refine management options and strategies generally or on a case-by-case basis.

Qualitative surveys of seabed substrates for the presence of naturally deposited sediments, and the presence and health of CCA, at both potential impact and reference monitoring sites during relevant spawning seasons can help understand the typical extent of natural pressures operating and provide context for evaluating the extent of any additional pressures that might be attributed to dredging. Quantitative data would be preferable, but the inherent difficulty (and cost) in obtaining and interpreting the data may outweigh the potential benefits.

# 3 Fishes

Knowledge about the effects of dredging on finfish is not as well understood as for sessile benthic communities, and information on spawning times and locations is sparse with limited published scientific data. The DSN undertook a program to assess the current state of knowledge regarding the effects of dredging-related 'pressure' on key finfish reproductive processes generally, and to collate the current knowledge of the key ecological windows around spawning and recruitment for Western Australian fish species. This was done to provide proponents with the opportunity to avoid and minimise dredging related pressures and impacts. The program was desk-based and involved reviewing the relevant papers in scientific journals and 'grey' literature, and also considering unpublished information and advice from experts in the Western Australian Department of Fisheries (now DPIRD) and research organisations and universities (see Harvey et al 2017).

## 3.1 Fish reproduction patterns in Western Australia

Data on the timing of spawning were only available for 56 species of the 102 species of fish that were deemed to be commercially and recreationally important in WA (see Harvey et al 2017 for data on each species). Spawning occurred throughout the year with the main spawning period between October and April in both tropical and temperate species although seasonality was more pronounced in the tropics with up to 80% participation (Figure B1).

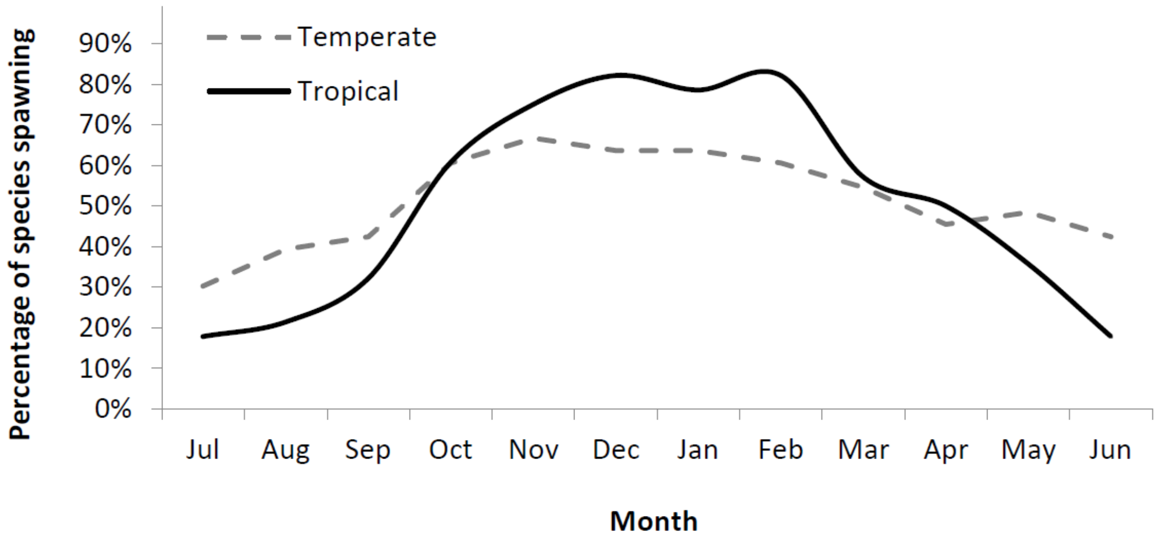


**Figure B1: Known spawning periods for tropical and temperate fish species caught and retained by commercial and recreational fishers in Western Australia (n=56; from Harvey et al 2017)**

Spatial and latitudinal variation in spawning period was evident for some species such as pink snapper (*Chrysophrys auratus*) leading to distinct stocks requiring individual management strategies (e.g. timing of fishing closures to protect breeding aggregations). The knowledge base for pink snapper breeding periods in bioregions from the Gascoyne to the South Coast is relatively well developed and summarised in Harvey et al (2017).

From the limited information available it is not considered advisable to rely on assumptions that spawning times for a particular species will be the same in different regions or even in different localities within a region.

There can also be spatial components with habitat types or specific locations of particular importance. For example, nearshore habitats (e.g. macroalgal beds) are critical for many emperor (*Lethrinidae*) species and Cockburn Sound is the major spawning area for pink snapper on the lower mid-west coast of WA.

If the potential threat to fishes is a key environmental issue in the EIA of a particular proposal, there may be a need to undertake specific research into spatial and temporal spawning patterns within an area of interest. Given the typical levels of fish species diversity it may be useful to first undertake a prioritisation exercise to rank species based on their economic and societal 'value', and their 'vulnerability' to dredging pressures which is based on biological aspects such as their reproductive strategies, degree of habitat specificity and distributions to ensure effort is appropriately targeted (see Harvey et al 2017 for further information).

# 4 Other marine biota

The WAMSI DSN found that locally relevant information on the life histories of non-coral and non-fish marine biota (seagrass, macroalgae, sponges, ascidians, bryozoans, molluscs, echinoderms, crustaceans and non-coral cnidarians) that would inform management decisions is difficult to find and much remains unpublished (see Short et al 2017). It also found that information was lacking for a range of species of invertebrates, seagrasses, and macroalgae that are known or likely to be ecologically significant in WA.

Where suitable data were available, the life histories of non-coral and non-fish marine biota were identified and listed in detailed tables with specific reference to potential effects of dredging at each life history stage (Short et al 2017). The information in these tables provides a basis for identifying potential critical environmental windows of sensitivity and evaluating the degree of confidence that can be placed upon them.

In WA, it appears that many marine organisms are most vulnerable to disturbance during the late spring to early autumn period (Oct–April) due to the timing of sensitive life history periods (periods of reproduction and recruitment). So, in broad general terms, and where critical life history processes can be adversely impacted by dredging activities, winter is a period of the year when dredging would be likely to pose the lowest risk to critical life cycle processes for a number of taxa in WA.

These broad generalisations do not hold true for all taxa and in all regions, and in many cases there are insufficient life history data to identify particular windows of sensitivity.

A species' reproductive strategy, reproductive season and developmental strategy are major factors contributing to its vulnerability to stressors. Brooding species with a limited capacity for dispersal are generally more vulnerable than those with planktonic larval stages.

Organisms that have a single reproductive episode in a life-cycle would also be expected to be more vulnerable to a dredging event compared to those which may reproduce multiple times in a lifecycle. Similarly, the effects of dredging during reproductive periods would be expected to be more detrimental for species with a discrete annual spawning period compared to those with multiple protracted spawning events occurring throughout the year.

The key considerations and known timings of key lifecycle processes for invertebrates, seagrasses and macroalgae in WA are summarised below. The information is for a limited range of taxa, and where projects have the potential to impact commercially and/or recreationally important species such as crayfish, prawns and oysters, proponents should seek out and apply relevant information (if available) for these circumstances.

## 4.1 Invertebrates

In WA, most corals are known to spawn synchronously during a discrete and predictable annual window in autumn (see section 1.1). While there is little information available for other invertebrates, other phyla have also been observed spawning in concert with the corals during the annual autumn spawning events, including echinoderms (sea stars and urchins) and polychaete worms (*Eunice* spp.). In WA, polychaete spawning has been observed to occur synchronously (coincident with coral spawning) over 12 degrees of latitude from Dampier in the north to Rottnest Island in the south.

Although limited, and opportunistic in nature, these observations suggest that autumn may be a period of particular importance in the life cycle of a range of tropical marine invertebrate species in north-western WA Furthermore, the autumn spawning period coincides with the onset of the seasonal Leeuwin Current that transports warm, tropical water (including larvae) southwards along the continental shelf break during autumn/winter and is largely responsible for sustaining Ningaloo Reef and the high latitude coral reefs of the Abrolhos Islands. If natural oceanographic processes along the WA coast have led to the preferential selection and establishment of tropical biota that have a genetic legacy to spawn in autumn, it could be expected that other tropical marine taxa found on these reefs, particularly those that only spawn annually, may reproduce during this period as well.

For these reasons, environmental windows established to reduce dredging related turbidity generation (and sediment deposition) around the neap tide periods in autumn (e.g. to protect corals) would likely reduce risks to sensitive life stages of other invertebrate taxa associated with coral reefs in north west WA. It should be noted that coral larval settlement is demonstrably sensitive to levels of deposited sediments on surfaces and it is thought that this may also be the case for other sessile invertebrate species.

## 4.2 Seagrasses

Seagrasses can be grouped into three broad categories (i.e. colonising, persistent and opportunistic) that reflect their reproductive, dispersal and growth strategies.

Colonising species (e.g. *Halophila* spp.; *Halodule* spp.) have short ramet turnover times, are quick to reach sexual maturity and allocate a significant amount of energy into sexual reproduction to produce seeds, usually resulting in the presence of a seed bank.

In the wet-tropical Kimberley region, the lifecycle of *Halophila decipiens* follows light availability in deeper water habitats, with seed dispersal during the light-poor wet season, and seedling growth, meadow development and gamete production occurring during the dry season when water clarity and associated light availability is high. Dredging activities during the dry season in the Kimberley region would place the greatest pressure on this species as the plants rely on higher light levels to stimulate germination of the seed bank, growth and meadow development and gamete production.

The seasonal growth and reproductive patterns of colonising seagrasses in the Pilbara are spatially and temporally variable and so no definitive and generally applicable environmental window of sensitivity can be identified at this stage.

Persistent species (e.g. *Posidonia* spp.) have long turnover times, can contain significant energy stores, are slow to reach sexual maturity and place less investment in sexual reproduction with seed banks rarely present. As such this group is more resistant to disturbance but takes longer to recover than colonising species.

The focus for management in temperate/sub-tropical regions where these meadows dominate is to reduce pressure during the summer months to increase flowering and fruiting success and to allow carbohydrates to be generated and stored to support seagrass survival during winter.

Opportunistic species (e.g. *Amphibolis* spp.; *Zostera* spp.) share traits with species from both of the previous classifications, with the ability to colonise quickly, produce seeds and to recover from seed when necessary. In WA, *Amphibolis* species flower during autumn, with gametogenesis occurring between May and October. The seed germinates on the adult plant and is released as a mature seedling between November and June and seedlings are present year-round. Therefore, it is possible that dredging in the months leading up to flowering (i.e. during autumn) could reduce carbohydrate reserves and flowering and hence reproductive capability.

Avoidance of dredging during the warmer months is likely to be beneficial to the reproductive success of *Zostera* species, while avoidance of dredging during the autumn will be beneficial for *Amphibolis* species.

## 4.3 Macroalgae

As with seagrasses, environmental windows for macroalgae should account for plant phenology, sensitive periods in the life history cycle (e.g. gametophyte vs. sporophyte stages for some macroalgae) as well as annual cycles in environmental conditions. *Sargassum* spp. and kelp (*Ecklonia radiata*) are the dominant canopy forming algae in WA.

The most common phenology of *Sargassum* spp. in temperate WA appears to be characterised by a spring-summer growth period, followed by reproduction in late summer, then senescence. This pattern may not apply to tropical populations.

Production of zoospores by *E. radiata* in temperate habitats is seasonal, primarily occurring from early summer to autumn (Dec–May), with a peak in April. Winter is the season of slowest growth, and significant thallus erosion and dislodgement due to storm conditions.

The phenology of most green and red algae (*Chlorophyta* and *Rhodophyta*) is unknown and generalities with respect to these groups cannot be made at this stage.

Based on a vulnerability assessment and the known timing of reproduction and recruitment for these canopy-forming macro-algal groups in temperate waters of WA, dredging would pose the lowest risk during August–September, when neither of the major habitat forming macroalgae are undergoing reproduction or recruitment.

# 5 Addressing information gaps

Given the paucity of information on life history characteristics and vulnerability to dredging for most marine species in WA, any management that considers environmental windows should ideally be based on locally-derived information rather than on generalities or information from other regions.

Where practicable, baseline studies to identify relevant environmental windows of sensitivity should also be designed to determine whether the environmental windows have a spatial component, for example specific spawning aggregation areas or substrate types for larval settlement. The vulnerability assessment undertaken in Short et al. (2017) could be used to identify those taxa most likely to be vulnerable to dredging pressures and hence to focus investigations.

Any data gathered on life cycle processes through the EIA of dredging proposals, and through other activities, should be captured and made generally available via IMSA to inform management and decision making in the future. The compendium of tables generated through DSN research (described above) could be reviewed and updated, and new tables generated, as new information comes to hand.

# Appendix C: Dredging-related environmental surveys, monitoring and management

## Contents

# 1 Introduction and scope

Dredging Environmental Monitoring and Management Plans (DEMMPs) are used to ensure dredging and related activities are adequately managed so that the extent, severity and duration of impacts are kept as low as reasonably practicable and the Environmental Protection Outcomes (EPOs) established through Ministerial Conditions of approval are achieved (see section 3.6.1 of the main body of this technical guidance).

The EPA expects proponents to prepare draft DEMMPs as part of the documentation submitted in support of the environmental impact assessment process. These are linked to the prediction of impacts and provide the EPA with a basis to understand how, and the extent to which, proponents intend to manage their activities. The relative levels of confidence in impact predictions, and in the efficacy of proposed management, are major considerations of the EPA when it prepares advice to the Minister for the Environment on whether the overall proposal is likely to meet the EPA's objectives for each relevant environmental factor. To that end, DEMMPs need to be clearly set out, with specific objectives linked to proponent commitments and the conditions of approval[13]. They need be underpinned by targeted monitoring programs, utilising scientifically defensible indicators and trigger values, and contain unambiguous assessment methods and reporting protocols.

This appendix has been prepared to assist proponents to develop efficient and effective work programs to support the environmental impact assessment and management (EIAM) of their dredging projects. Although it's primary focus is on dredging management, impact assessment and management are interlinked and have overlapping data requirements. As such, this appendix is structured according to key steps in the EIAM processes, including conducting pre-development surveys to characterise the environment that may be influenced by implementation of the proposal to support the preparation of proponent's environmental impact assessments and proposed management, and post-approval monitoring and management.

The content has been developed in consideration of the results of the WAMSI DSN coupled with the practical experience of EPA advisors, environmental consultants and proponents in the assessment, management and regulation of a number of large-scale dredging programs in north-west WA.

General advice is provided in section 2 on characterising the physical environment and establishing baseline conditions for water quality and benthic light availability.

Advice on characterising aspects of the biological environment, including collecting relevant information to support environmental impact assessment and identifying possible bioindicators of stress and impact to support management and inference assessments, is provided in section 3.

Guidance for developing DEMMPs, including approaches for deriving guideline trigger values that are linked to the impact predictions and that could be used in a tiered management system for protection of the key benthic communities likely to be affected by the dredging program, is provided in section 4.

The advice and approaches set out here are designed to be consistent with, and complement, the framework set out in the main body of this Technical Guidance. The overarching intent is to improve confidence and reduce environmental risk associated with the implementation of dredging programs and to facilitate continual improvement in the EIAM of future dredging projects. Proponents are reminded that the advice provided here may not be appropriate in all circumstances and they are encouraged to seek advice from the EPA Services Directorate regarding the application of the guidance contained in this appendix, including their proposed DEMMPs, in the context of their proposals.

[13] When preparing draft DEMMPs for EIA purposes, proponents should include 'likely' conditions of approval developed in consultation with the EPA Service Directorate.

# 2 Characterising the physical environment

This section provides generic advice on establishing environmental baselines for relevant water quality and sediment parameters to support the assessment and management of dredging-related impacts on all benthic community types. There is significant overlap between the data sets required for ecological impact assessment and for dredge plume modelling. Proponents should consider the guidance on establishing relevant baseline conditions and associated data sets to support sediment transport and fate modelling provided in Sun et al (2020) as well as the baseline data sets required to address the impact of dredging on other environmental values before designing environmental baseline monitoring programs. There are likely to be efficiencies to be gained by considering the respective data acquisition requirements in combination, identifying overlaps, and designing and implementing data capture programs concurrently.

## 2.1 Baseline conditions

Pre-development surveys should be used to characterise relevant physical characteristics of the environment that can be affected by the dredging operation and how these characteristics vary: spatially (e.g. nearshore vs offshore, exposed vs. sheltered); in response to different energy regimes (e.g. calms or storms); and temporally considering seasonality and inter-annual variability. The dominant pressure-response pathways, that link sediments suspended by dredging to the local biota, are of particular importance.

The key water quality parameters to measure are total suspended sediments (TSS), turbidity as NTU, and seabed light measured as photosynthetically active radiation (PAR) used to derive benthic daily light integral (DLI) (see section 2.1.1). For a given amount of solar radiation at the sea surface, the depth and attenuating properties of the water column control the quantity (and quality) of light reaching the seabed. Many of the guidelines for impact prediction in Appendix A have seabed light intensity components expressed as DLI.

Suspended sediments, organic particles, phytoplankton and water colour influence the attenuation of light (measured as light attenuation coefficient; LAC) through the water column to varying degrees. Near dredging locations, the major controlling factor on LAC will become the sediments liberated to the water column. In most cases understanding the generation and transport of sediments, and the subsequent effects on seabed light availability, will be important for impact prediction purposes.

Timeseries data should be collected for key parameters to capture the typical range of conditions across the area that may be influenced by dredging. Particular emphasis should be on the areas where benthic communities are present (e.g. coral reefs, seagrass meadows, sponge gardens). Concurrent measurements of the key parameters (i.e. TSS-NTU-LAC) should be taken where possible to establish relationships necessary for impact prediction and management. Recent advancements in remote sensing analysis techniques allow historical water quality datasets to be generated from archived satellite imagery in some cases (see section 3.1.2 of the main body of this technical guidance; Fearns et al 2019). This capability can allow seasonal patterns and interannual variability of certain water quality parameters to be examined under a range of climatologies.

Where the intention is to use thresholds derived relative to baseline conditions for impact prediction and/or management, long time-series (ideally ≥2 years) of relevant environmental quality data should ideally be collected to ensure the full natural exposure regime is adequately captured. Data affected by cyclones in the baseline phase need to be examined on a case-by-case basis, particularly if baseline data are used for deriving thresholds or management triggers and the extreme percentiles (e.g. $P_{95}$, $P_{99}$; $P_{05}$, $P_{01}$) are used (see section 3.2.1 and Jones et al 2019a for more information). The key consideration for inclusion/exclusion of such data is whether it is reasonable to assume that cyclones have had extreme effects on the baseline, based on the data collected and also on the size, intensity and proximity of the cyclone (see ANZWQG 2018, EPA 2016a, EPA 2017 for more information).

Ambient water temperature is an important factor controlling the distribution and health of marine communities. Regional ocean warming and marine heatwaves are becoming more frequent due to climate change and even rises of only a few degrees can cause extreme stress and even change community structure (Wernberg et al 2016). Coral bleaching is often associated with elevated seawater temperatures and it has been established that bleached corals are particularly vulnerable to elevated levels of sediment deposition due to the metabolic cost of actively removing sediment and inability of bleached corals to replenish energy reserves through photosynthesis.

Elevated seawater temperature events during dredging can confound the interpretation of dredging environmental monitoring data. Although water temperature is not affected by dredging, for the reasons outlined above it is a useful parameter to measure to establish baselines during 'normal' conditions and under abnormal (e.g. heatwave) conditions, before and during dredging. Furthermore, seawater temperatures can be forecast in advance and in some cases knowledge of the timing of these events can be used to inform the scheduling of dredging to avoid or reduce additional stress on sensitive benthic communities during these periods.

### 2.1.1 Light

Arguably the most important parameter to measure and characterize over the annual cycle is photosynthetically active radiation (PAR) availability at the seabed. PAR intensity is expressed in terms of photons of light per unit area and per unit time. Instantaneous measurements of PAR typically have the units of umol photons $m^{-2}s^{-1}$. The Daily Light Integral (DLI) is the total PAR received by benthic communities over a single day (i.e. mol photons $m^{-2}d^{-1}$) and underpins many of the suggested guidelines for predicting impacts to corals, seagrasses and some sponges (see Appendix A). Even if water clarity remained unchanged seasonally, there is strong seasonality in the DLI due to changing day-length and solar angle. Other seasonal factors such as prevailing wind speed and direction (and associated wave heights and fetch distances) increase or decrease the re-suspension of sediments and are superimposed on the seasonal daylength cycle. Important points to consider include:

- The PAR regime at the seabed at both potential impact and reference monitoring sites should be characterized through at least one full annual cycle (or at least for each season) using underwater sensors (with automated wipers on sensors) and data loggers.
- Water quality parameters should be measured at the site regularly, at least during every logger service, to provide contemporaneous measurement points for related parameters (e.g. SSC, stratification, light attenuation coefficients) to assist in modeling seabed light.
- It is useful to co-locate turbidity and PAR loggers where practical to provide further validation data for predictive algorithms and establishing management trigger values.
- Relationships between natural TSS and LAC should be established. This may require a dedicated campaign which if executed during a period where energy levels range from low to high (e.g. period leading up to and including a significant swell event), would provide useful relationships over a short period to assist in modeling 'background' conditions.
- Relationships between 'likely dredge-generated' TSS and LAC should be established wherever possible. This is not possible pre-dredging but laboratory-based approaches can be used to develop relationships for a range of particle size classes likely to be encountered or generated during dredging (see Fearns et al 2017). Sediments are suspended in a vertical column and allowed to settle leaving increasingly finer sediments in suspension over time. Time-series measurements of TSS and LAC within the column provide relationships for a range of suspended sediment size classes.
- Data on incident PAR at the water surface, that captures the effects of cloud, solar angle and local atmospheric conditions, will be required to interpret benthic light data and to derive LAC from seabed loggers. These data may be available from meteorological stations run by the Bureau of Meteorology or other providers if nearby, otherwise a dedicated meteorological station(s) will need

to be established in close proximity to the study area (see Sun et al 2020) for other meteorological data relevant to plume generation and fate modeling).

- Remote sensing algorithms are available for retrospective analysis of the PAR climatology over a number of years to help understand and characterize broad spatial patterns and the inter-annual variability in them, but their utility is related to the extent of calibration of algorithms used for this purpose (see below).

### 2.1.2 Suspended sediment

Understanding the natural levels and characteristics (e.g. particle size distributions; mineralogy; organic content) of sediments suspended in the water column may be important for characterising natural ambient conditions and for modeling the additional effects of dredge-generated suspended sediment concentrations, particularly the consequences for seabed PAR. It is suggested that:

- Water samples should be collected *in situ* to characterise particle size distributions; organic content, mineralogy (e.g. silicoclastic, carbonate) of sediments suspended in the water column.
- Knowledge of the energy levels in the water column and atmosphere (i.e. the metocean conditions) prior to, and at the time of sampling would be very useful for predictive purposes (typically obtained from wave and current meters and meteorological stations).
- It would be prudent to coincide water quality characterisation surveys with cloud-free satellite overpasses where possible and to capture surface reflectance (upwelling radiation) and other data to assist in tuning algorithms to convert satellite generated reflectance data to the ecologically-relevant parameters such as TSS and seabed PAR levels (Fearns et al 2019). Fearns et al (2019) provides comparative analyses of a range of published algorithms, and makes recommendations on those that are most suitable for use in the Pilbara.

### 2.1.3 Deposited sediment

Deposited sediments, and sediment deposition rates, are difficult parameters to measure and describe in ecologically meaningful terms[14]. Sediment traps and deposition sensors have been used to 'measure' deposition but the results are relative and not absolute, and there is still considerable uncertainty associated with the interpretation of such data. The characteristics of undisturbed surficial sediments can be used to infer the extent of any sediment deposition associated with dredging activities (e.g. before and after), and the persistence of any changes (e.g. repeat surveys over time). Although sediment deposition data are not recommended to be used directly for management or stand-alone compliance assessment, these data can be useful to calibrate sediment deposition/re-suspension models and provide a line of evidence to support inference assessments associated with management or compliance reporting. If this is the case then:

- Un-disturbed surficial sediment samples (e.g. surface 10 cm) should be collected proximal to, and along transects away from, the area likely to experience elevated sediment deposition (e.g. transects perpendicular to channel orientation; radiating from berth pockets) with sample density decreasing with increasing distance from dredging activity[15]. Sites should be geo-referenced to allow repeated samples during and post dredging if required.
- Sediment particle size and type should be characterized to allow assessment of the silt/clay, fine sand and coarse sand fractions for different depth intervals down the 10 cm profile. Bulk density of each sediment fraction should be determined to assist in mass-balance assessments of dredged sediment fate (in terms of percentage of volume and weight of material dredged).

[14] See Jones et al 2019a for discussion of this issue.

[15] See EPA 2016b for advice on sediment sampling.

### 2.1.4 Existing seabed characteristics

- Understanding sediment dynamics (deposition, re-suspension) and characteristics (e.g. organic matter content; particle size distribution) of the seabed can provide insight into natural levels of pressure (e.g. stability) and context for developing DEMMPs.
- Seabed morphology (e.g. presence/absence, size and orientation of sand ripples) is a useful indicator of relative sediment stability/dynamics.

# 3 Characterising the biological environment

This section provides advice on undertaking surveys of benthic communities and habitats in north-west WA across the area that may be affected by a dredging proposal. These surveys are typically undertaken for a range of purposes to support EIAM including to identify, characterise and map the key BCH present and assess their relative baseline health status. The results are used for impact prediction and also to support environmental monitoring and assessments for management and compliance purposes.

It is acknowledged that field surveys are often logistically complex and costly to undertake. The advice in this section is provided to help proponents design and execute field programs that are efficient and effective. It includes considerations and advice for undertaking pre-development surveys, habitat mapping and identifying possible bioindicators of benthic community health. Although the advice is presented separately for corals, seagrass and sponges (filter feeders), there are obvious areas of complementarity. Proponents are encouraged to consider the advice for the functional group(s) relevant to their dredging locations and circumstance, and design and execute their field programs accordingly.

## 3.1 Corals

### 3.1.1 Pre-development surveys and baseline conditions

#### Locating and characterising coral communities

Given that light reduction is a key dredging-related pressure for coral, and for a given level of water clarity light availability will attenuate with depth, it is important to characterise the existing coral community across the full range of depths of coral habitat that may be affected by dredging pressures. Ideally, the abundance and/or occurrence of key target taxa within the coral communities should be monitored through time to properly quantify natural variability in the event that a before/after control/impact (BACI) design will be used to assess and manage dredging related impacts. This is particularly important for long-duration and large-scale dredging programs.

Initial surveys should be designed to maximise the likelihood of identifying coral communities and habitats by taking account of habitat preferences and physiological requirements. Issues that should be considered include:

- Typical coral habitats are hard substrates where water clarity and depth are such that there is sufficient PAR at the seabed to allow photosynthesis of the algal symbionts. Aerial imagery and bathymetric charts are useful for identifying possible coral habitat and to focus survey effort. Surveys should target high- and low-relief reefs or pavements where water clarity and depth provide at least 5% of surface irradiance at the seabed (measured as PAR). Light limitation generally restricts coral habitat to water depths of less than 15 m in nearshore coastal waters, but coral habitat can extend to depths of 50 m in clear oceanic waters outside of coastal influences (e.g. Scott Reef).

- Surveys should target moderate energy environments characterised by sediment PSDs skewed to the coarser end. Coral reef development is less likely to occur in areas characterised by fine, readily resuspended sediments, or areas receiving significant river runoff (e.g. adjacent to river mouths) even if rainfall and runoff is infrequent or episodic (e.g. the dry tropical Pilbara region).
- Individual corals can be found within algal reef communities and the shallower filter feeding communities because of similar habitat requirements. Some species can tolerate periodic exposure to air and hence can be found in the intertidal zone. Reef growth is restricted vertically by exposure and will not generally occur above neap high water levels.
- All corals are perennial so seasonality of occurrence is not a consideration for field surveys. However, the marked seasonality and synchrony in coral reproduction in WA is well known (see Appendix B) and it is useful to understand the specific timing for the dredging area to allow management to be tailored accordingly. In the interests of efficiency, consideration should be given to planning coral surveys to occur immediately before these potential spawning periods so that collections can be made to assess most likely spawning timing in the vicinity of the development proposal (see section 1.1.1 of Appendix B and references therein for specific guidance).
- Surveys should record depth and time (to provide a reference to chart datum) and substrate type, exposure and orientation to prevailing swell waves (useful for habitat modelling).
- Observations of recognised bioindicators of coral health (see section 4.2.2) such as bleaching, deposited sediments and recent mortality, and also predation scars and coral predators, should be recorded. Habitat utilisation information, particularly habitat-linked species such as demersal fishes, should also be recorded.
- It can be useful to undertake a census of coral size class frequency (or to identify sites that would be suitable for this purpose). The numbers of individuals within defined size classes, for each morphology type, coupled with knowledge of specific growth rates can be used to assess the degree of inter-annual stability and levels of recruitment in the population (see section 3.1.4).

### 3.1.2 Coral habitat mapping

The relatively persistent nature of corals coupled with the link to hard substrates and tendency to be found in relatively clear waters means that in general coral reef habitat is reasonably straightforward to map. Some issues to consider include:

- High quality remote sensed products such as aerial photography (preferred) and some satellite imagery can be used to delineate features that may be coral reefs. These areas can be targeted during field surveys. It is important to ensure that all decision rules used to interpret the imagery and create habitat maps are clearly set out. If photography is specifically taken for this purpose then try and coincide with low tide and/or neap tidal cycles when waters are typically clearer, and an oblique sun angle (before 1000 h or after 1400 h) to reduce sun glint, and use yellow and polarised filters to increase contrast to better delineate benthic features.
- Corals have physiological requirements (light and temperature), morphological characteristics (range of morphologies from branching to massive, with varying degrees of tolerance to moderate-high wave energy) and other requirements (e.g. type of substrate) that together define their habitat preferences. There are species-specific habitats with erect branching species typically found in more sheltered areas; massive species in more exposed areas.
- It should be noted that coral habitat can sometimes have little or no live coral growing on it because of previous disturbance, so areas with live coral and areas that have characteristics that suggest they should support coral should be identified. Note that it is particularly important to identify coral habitat in and adjacent to areas that are likely to be subject to direct disturbance (e.g. channels, breakwaters, spoil grounds) as this information will be required to determine the extent of any permanent impacts (i.e. > 5 years to recover) and recoverable impacts for addressing the requirements of EPA (2016a).

- Ensure all decision rules for assigning habitat categories are clearly set out. When coral is a component of a mixed community (e.g. mixed filter feeder and coral) the habitat description architecture should be configured so that it can be interrogated to identify habitat that supports coral and would display as part of an overall coral layer. This is required for assessing cumulative impact and loss of 'coral' habitat (and other benthic communities and habitats) within Local Assessment Units (LAUs) as set out in EPA (2016a).

### 3.1.3 Selecting coral health bioindicators

Assessing bioindicators generally requires diver or diver-less surveys to capture measurements or images and as such are not amenable to automated monitoring (i.e. compared to water quality parameters). However, they are direct measures of coral health and useful for assessing the current status of a site prior to development and disentangling natural and dredging-attributable effects and impacts during dredging programs. Bioindicators can form a component of adaptive management programs and can also inform compliance assessments because of the added confidence they bring to decision-making (see section 4). As such, it is useful to gather as much relevant information as possible in the pre-development phase to better understand the natural levels of stress (and associated pressure pathways) to inform impact predictions and for potential use for management purposes on implementation of the proposal. Some possible bioindicators for coral health at the pre-development phase, and the rationale for their selection, are outlined below.

*Sentinel species*

- The branching species *Pocillopora damicornis* and *Acropra millepora* are tolerant of sediment deposition but particularly susceptible to light reduction from high SSC. These species are relatively easy to identify in the field and locations of colonies should be recorded for use as sentinels for identifying light stress if bioindicators form a component of adaptive management programs. The presence of these species also indicates a 'high light' environment which is useful to know for impact prediction purposes.
- The foliose (e.g. *Montipora* spp.) and massive species (e.g. *Porities* spp.) are more tolerant of reduced light but far more sensitive to deposition than the branching species. They are relatively easy to identify in the field and locations of colonies should be recorded for potential use as sentinels. Under low energy conditions, the presence of these species and the absence of the branching morphologies may also be used to infer that the area is subject to chronic or periodic turbidity events, which is useful for impact prediction purposes.

*Bleaching and accumulated sediment*

Coral bleaching reduces the ability for corals to obtain energy from photosynthetic pathways (via the algal symbionts) and is becoming increasingly prevalent globally as oceans warm. DSN research has demonstrated that bleached corals are largely unable to actively remove sediments from their surfaces which makes them much more vulnerable to sediment smothering; a finding that is useful for impact prediction and also for developing dredging management programs (Bessell-Browne et al 2017a). Corals can also bleach when subjected to chronic light stress and/or deposited sediment stress so it is prudent to capture pre-dredging information on natural levels of coral discolouration and bleaching at sites potentially affected by dredging and at reference sites. This information can assist in interpreting the results of monitoring programs.

- Baseline surveys should record the extent of the natural levels of coral discoloration or bleaching that may be evident. This information can be compared against the antecedent water temperatures (see section 2.1) to identify critical temperature regimes that may promote bleaching. This information could be used in management plans to allow additional management measures to be put in place when these conditions occur, to reduce dredging related pressure to below that which would be normally tolerated by healthy (un-bleached) corals.

*Mucus sheet formation*

Mucous sheet formation in *Porites* spp. colonies is a sub-lethal response to deposited sediment (Bessell-Browne et al 2017b).

- Baseline surveys should record the presence of any mucus sheets on *Porities* spp. and the percent coverage of the surface of affected colonies. The number of sheets produced over time by individual colonies can provide knowledge of the natural periodicity of exposure to sediments, and the conditions that induced sheet formation, but would require numerous repeat surveys.

### 3.1.4 Assessing recovery potential and pathways

The composition and cover of a coral community at any point in time is in a state of flux; reflecting the net result of the rates of growth and mortality of individuals within the population, and levels of recruitment to the population. Mortality occurs naturally from a range of causes including predation and storm damage. Corals can recover from impact by growth of remaining colonies or by establishment of new recruits.

Coral demographic data (number of individuals in each predetermined size class) can be used to describe the community but importantly, with two or more years of data, it can also be used to construct population models and make projections of likely impact and recovery scenarios tuned to local species and conditions (Babcock et al 2017). It can also be used evaluate the consequences of cyclones and other natural disturbances that may occur from time to time. If coral population models are to be constructed then consult Babcock et al (2017) for detailed methods and advice. Key considerations include:

- Re-locatable transects should be established in areas that have significant numbers of individuals, across a range of size classes, of the target species. Size classes should be selected (e.g. 5 - <15 cm, 15 - <25 cm, etc.) and each colony within each size class should be measured and tagged so that it can be re-located on the next sampling occasion.
- Approximately 50 individuals within each size class, spanning a range of sizes within that size class, is generally required to achieve a statistical power of 0.8 and effect size of 0.05. At least 2 years of pre-impact data are required.
- Recruitment data are critical, particularly for *Acropora* spp., and hence the quality and reliability of inferences made will improve with increasing numbers of years of pre-impact data on levels of recruitment to natural substrates (i.e. to establish the baseline condition).
- It is important to ensure sampling is seasonally consistent between years and is best undertaken between 12 and 18 months after the spawning date of recruits that are considered most likely to be detected.

## 3.2 Seagrasses

### 3.2.1 Pre-development surveys

Pre-development surveys offer the opportunity to gain an understanding of the seagrass assemblages and associated baseline water quality and sediment characteristics, at potential impact monitoring and reference sites. The most important set of baseline data to collect for seagrasses relates to light quantity. Obtaining continuous light data over an annual cycle (or preferably more) of pre-development conditions at the seabed would provide a valuable baseline of the frequency and duration of light conditions that are suitable for the seagrasses at that site and can form a reference condition for impact assessment modeling (see section 2.1 above. for general advice).

*Locating and characterising seagrass*

- Seagrasses are only found where there is sufficient light at the seabed to meet minimum requirements for maintaining a positive carbon balance and where sediments are relatively stable. They can occur as mono-specific or multi-specific meadows or as part of a mixed assemblage within filter feeding communities. Some species tolerate exposure to air and occur intertidally. In these areas they often display dwarfism with miniature growth forms of *Halophila* spp. and *Halodule* spp. sometimes present, and if so, surveys are best undertaken at low tide on foot. This knowledge of physiology, coupled with bathymetry data and an understanding of natural levels of water clarity (e.g. light attenuation coefficients), can be used to identify possible seagrass habitat and provide a basis for initial survey design.
- Pre-development surveys provide the opportunity to identify the composition of mixed-species seagrass communities to improve confidence in threshold selection/development and impact prediction, selecting relevant bioindicators and assist in designing appropriate monitoring programs.
- If guidelines for impact prediction and management of a mixed assemblage of seagrasses are based on pressure-response data for a single species within the assemblage, it is important to focus on the species that is most sensitive to the relevant pressure (e.g. reduced PAR availability) to avoid under-estimating the impact on the assemblage as a whole.
- The species that could be encountered in NW WA and their broad habitat preferences are presented in Table C1 below:

**Table C1: Seagrass species and their habitats in NW WA (from Lavery et al 2018)**

| Species | Clear | Turbid | Intertidal | Subtidal | Estuarine | Coastal | Reef | Deep | TOTAL |
|---|---|---|---|---|---|---|---|---|---|
| *Hydrocharitaceae* | | | | | | | | | |
| *Enhalus acoroides* | X | X | X | X | X | X | X | | 7 |
| *Halophila decipiens* | X | X | | X | | X | X | X | 6 |
| *Halophila ovalis*[1] | X | X | X | X | X | X | X | X | 8 |
| *Halophila spinulosa* | X | X | | X | X | X | | X | 6 |
| *Thalassia hemprichii* | X | X | X | X | | X | X | | 6 |
| *Cymodoceaeceae* | | | | | | | | | |
| *Cymodocea angustata* | X | X | X | X | X | X | X | | 7 |
| *Cymodocea rotundata* | X | X | X | X | | X | X | | 6 |
| *Cymodocea serrulata* | X | X | X | X | X | X | X | | 7 |
| *Halodule uninervis*[2] | X | X | X | X | X | X | X | | 7 |
| *Syringodium isoetifolium* | X | X | | X | X | X | X | | 6 |
| *Thalassodendron ciliatum* | X | | | X | | | X | | 3 |

[1] including *Halophila ovata*, minor

[2] including *Halodule pinifolia*

- Site-specific surveys would be required to characterize the seagrasses likely to occur at any given site, ideally conducted on several occasions. While seagrass cover in the Pilbara is highly variable over time, the likelihood of detecting seagrass, as well as flowering, fruiting and seed banks, is greatest if surveys are undertaken between November and February. In contrast, peak biomass of the deepwater *Halophila* spp. meadows in the Kimberley is most likely to be encountered towards the end of the dry season before they flower, set seed and senesce (Masini et al 2009). Seagrass is less likely to be found in the Kimberley during the wet season and early dry season (December to April).
- Disturbance history should be considered when selecting pre-development or compliance monitoring survey sites. Stochastic events (e.g. cyclones) appear to be important drivers of seagrass dynamics in the region, with changes in composition and abundance occurring over 2-5 years. Knowing the point in a longer-term recovery pattern that a site is at will allow more meaningful comparisons against, for example, post-dredging recovery monitoring data. In cyclone-affected areas, repeating surveys over several years will increase the probability of detecting seagrass.
- Because appropriate pre-development surveys will involve significant effort and resources, broad-brush pilot surveys would be useful to identify appropriate areas for more detailed survey and fine tune methods to local conditions.
- The presence of dugong can indicate that seagrass is nearby. Evidence of dugong feeding scars in seagrass meadows should be recorded as a record of habitat utilisation and to help understand levels of natural disturbance.
- Even when seagrass is present it can be hard to detect in tropical waters. Remotely deployed towed video or still camera methods might not be appropriate for detecting seagrass in the Pilbara, especially if towed relatively fast and/or significantly above the seafloor. Towed video cameras can be periodically dropped to the seafloor and kept motionless to help identify the presence of seagrass, particularly if *Halophila* spp. is expected.
- Diver operated still camera images taken within 1 m of the seabed are the most effective means of detecting and surveying seagrass. Gentle fanning of surface sediments can expose rhizomes and assist in detecting seagrasses.
- Grab samples of seagrass should be taken where possible (e.g. using mini-grapples dragged along the seabed) to identify species present and assess reproductive status.

### 3.2.2 Seagrass habitat mapping

- Satellite or airborne remote sensing methods will not be useful tools for mapping or monitoring the distribution of seagrasses in the Pilbara, however they are very useful for mapping perennial seagrasses in sub-tropical/temperate areas.
- The term ‘meadow’ is not generally appropriate in tropical waters and very sparce cover can still be ecologically important.
- Criteria based on percent cover is not very useful for distinguishing between seagrass and non-seagrass habitat in most tropical communities. As described in section 3.2.1, tropical seagrass can be quite cryptic and difficult to identify using most standard towed video habitat mapping approaches and the ephemeral nature and high interannual variability of tropical seagrasses make it difficult to define precise seagrass area boundaries.
- Seagrass ‘habitat’ can be more readily identified based on knowledge of the physiological requirements (e.g. minimum light, substrate type), and determining where these requirements are met in the area of interest. Historical data should be accessed where available to assist in defining seagrass habitat. That habitat can be separated into vegetated and non-vegetated (at the time of survey).
- Habitats can be further separated into those with demonstrable recovery potential (e.g. viable seed

bank), and those where there is no demonstrable recovery potential, noting that assessments of recovery potential and recovery timeframes will need to be undertaken if any seagrass habitats are likely to be impacted by the dredging proposal (see section 4.1.5).

- Habitats can be further resolved according to seagrass species composition. This provides useful contextual information, particularly for selection and tailoring of guidelines for impact prediction purposes (see Appendix A).
- Ensure all decision rules for assigning any habitat categories are clearly set out. When seagrass is a component of a mixed community (e.g. mixed filter feeder and seagrass) the habitat description architecture should be configured so that it can be interrogated to identify habitat that supports seagrass and would display as part of an overall seagrass habitat layer. This is required for assessing cumulative impact and loss of 'seagrass' habitat (and other benthic communities and habitats) within LAUs as set out in EPA (2016a).

### 3.2.3 Selecting seagrass health bioindicators

For similar reasons to those set out previously for corals (see section 3.1.3), it is useful to gather as much relevant information as possible in the pre-development phase to better understand the natural levels of stress (and associated pressure pathways) in seagrass communities to inform impact predictions and for potential use for management purposes on implementation of the proposal. Some possible bioindicators of seagrass health at the pre-development stage, and the rationale for their selection, are outlined below.

- Characterisation of seagrass morphological and physiological/biochemical responses to previous sedimentation and light history stress (e.g. vertical rhizome elongation and rhizome carbohydrate concentrations) can help refine appropriate thresholds to apply in impact prediction and management. For example, vertical rhizome growth is a reliable indicator of burial stress, but only in situations of high light and low organic content sediments (see section 4.2.2).
- Understanding the natural variability of key characteristics of seagrasses that respond to light availability at the plant and community levels in the locality and region is integral to the development of light-reduction impact thresholds when using the approach recommended by ANZWQG (2018).
- Pre-development surveys can be used to characterise the background natural variability of bioindicators of seagrass health at both reference and impact sites. This understanding could be used to refine thresholds for these bioindicators and avoid over- or under-prediction of impacts which could result in a loss of species diversity and ecological function or unnecessary time, effort and financial costs to complete a dredging operation.

### 3.2.4 Assessing recovery potential and pathways

- Knowledge of the genetic characteristics of seagrass communities in an area likely to experience dredging-related pressures can be useful for identifying and assessing the potential risk that dredging poses to them, and the likely recovery potential and pathways if impacts were to occur. The resilience of seagrass species to pressures can be influenced by the amount of genetic variation in that population. Seagrass meadows and species in NW WA display different levels of genetic diversity and so they may need to be managed differently. In addition to identifying which species of seagrass are present and likely to be influenced by dredging-related pressures, pre-development surveys could usefully collect data on:
  1. the genetic diversity of the seagrass meadows (i.e. clonal richness, allelic diversity, heterozygosity)
  2. the life-history traits relevant to potential recovery, e.g. sexual reproduction, seed banks
  3. the magnitude of gene flow and proximity of local populations (outside of the zone of impact) for re-population.

- If impacts are likely it will be important to confirm if and when flowering, fruiting and seed bank production occur. The presence (or absence) of seagrass seed banks in the sediment can help determine feasible recovery pathways and assess the likelihood of recovery following disturbance or loss which is particularly important for differentiating between the zones of High Impact and Moderate Impact (see section 3.2 of Appendix A). Sediment samples for determining the extent of any seed-banks can be readily collected while in the field and stored for subsequent analysis if required (see Vanderklift et al 2017 for methods).

## 3.3 Sponges

### 3.3.1 Pre-development surveys and baseline conditions

*Locating and characterising sponges*

Sponges can be found from the lower intertidal to beyond the continental shelf. They can occur on a range of substrates but they are primarily associated with relatively stable substrates. Taxonomic knowledge is poor for NW WA but DSN research has identified 1164 species and operational taxonomic units (OTUs[16]) in the Pilbara region alone, based on museum records and limited field sampling. The rate of endemism is high (over 20% in NWWA) and many species remain un-described.

A detailed colour image catalogue of filter feeders was developed during a field study at Onslow, which included sponges, hard corals, soft corals, gorgonians, ascidians, hydrozoans and bryozoans (Wahab et al. 2017b). The catalogue includes underwater (*in situ*) photographs and surface (above water) photographs, reference to the Western Australian Museum registration number for each specimen, and was designed as a practical resource for future marine environmental studies in the area. Information on chlorophyll-a concentrations and phototrophic capacities of sponges in the photo catalogue are included as Figure 7 in Wahab et al. (2017b). Some key issues to consider include:

- Proponents are encouraged to survey for filter feeding communities prior to dredging with a particular view to locating any that contain cup and phototrophic sponge species. These areas can be used to evaluate natural responses of sediment-sensitive species to ambient conditions (by comparison with physical water quality data). Depending on their locations relative to the impact and management zones associated with the proposal, they can be used to inform adaptive management and/or compliance monitoring.
- The photographic catalogue of sponges, and other benthic taxa, collected from the Onslow area (described above) should be used for standardisation of data with respect to sponge functional morphology and benthic taxa species identification where appropriate.
- Particular attention should be given to locating *Carteriospongia foliascens* which is widely distributed and demonstrably sensitive to dredging pressures. *Carteriospongia foliascens* and *Carteriospongia* spp., can be easily identified in high resolution photographs of the benthos from surveys using hand-held or drop down camera, towed video or ROVs.
- Habitat utilisation information, particularly habitat-linked species such as demersal fishes, should also be recorded.
- Remote sensing techniques are not generally utilised for identifying sponge communities. Simple single beam echo sounders can be used during field surveys to identify possible 'vegetated' areas for investigation and once calibrated can be used to undertake large numbers of transects to broadly delineate filter feeder habitats and communities and target suitable locations for quantitative surveys. Sea whips are particularly conspicuous and visible on echo traces as thin bent lines rising from the seabed.

*Baseline conditions*

Natural selection appears to play a particularly important role in determining the sponge taxa at a location and their levels of sensitivity to pressures such as those imparted by dredging. This selection could be based on chronic conditions (i.e. natural levels of turbidity, light availability, etc) or the previous history of acute events (e.g. intensity and frequency of cyclones). In naturally turbid areas, or areas that have a recent history of exposure to episodic events that generate intense turbidity, the conditions would favour (i.e. select for) sediment tolerant taxa and/ or morphologies less likely to accumulate deposited sediment. Conversely, areas with low levels of suspended sediment, and higher levels of ambient light at the seabed would 'allow' the sediment susceptible and phototrophic taxa to establish and grow. It follows that the sponge communities in these areas may be more susceptible to dredging pressures than sponge communities in turbid areas, which has implications for impact prediction and management. Other issues to consider include:

- The history of disturbance events such as bleaching events, cyclones and storms, and flooding events should be considered as part of the baseline habitat description phase. For cyclones, damage zone models exist that can predict whether a sea state is sufficient to severely damage benthic communities based on wind speed, duration and fetch (e.g. https://www.nature.com/articles/srep26009).
- Reference sites, least likely to be affected by dredging, should be identified and monitored prior to dredging to establish baseline conditions (e.g. species composition and indicators of sponge health such as degree of bleaching and necrosis) for these sites. These data can be examined in combination with data on the spatial patterns in historic cyclone exposures, to provide an insight into the trajectories of habitat responding from these types of conditions (e.g. recovery from cyclones).
- It is informative to evaluate natural turbidity, and sediment characteristics and dynamics of habitats that may be affected by the proposed dredging, to describe and understand typical conditions, and provide insight into the key drivers of community composition. To that end, it is useful to have relevant water quality data available (at least one year of pre-dredging water quality monitoring) to guide the first baseline assessments of filter-feeding communities. This will assist in the selection of reliable monitoring and reference sites that are equally affected by natural sediment dynamics.

### 3.3.2 Habitat mapping

Sponges are key components of most benthic filter feeding communities. This group has been singled out here because there is more knowledge of their susceptibility and resilience to dredging pressures than other filter feeding taxa and hence can be used to infer impacts to filter feeding communities more broadly. Hard corals that have calcium carbonate skeletons are also filter feeders but for the purposes of EIA they are considered and mapped separately (see EPA 2016a).

- Ensure all decision rules for assigning filter feeding habitat categories are clearly set out. When filter feeders such as sponges are components of mixed communities (e.g. mixed filter feeder and coral) the habitat description architecture should be configured so that it can be interrogated to identify habitat that supports filter feeders and would display as part of an overall filter feeder layer. This is required for assessing cumulative impact and loss of 'filter feeder' habitat (and other benthic communities and habitats) within LAUs as set out in EPA (2016a).
- Filter feeder habitats containing sponges can be usefully separated according to nutritional mode based on presence and absence of phototrophic sponges. Consideration could also be given to separating based on dominant morphologies (e.g. encrusting vs erect) and even within the erect morphologies (e.g. cup vs barrel). These are important considerations for identifying key pressure-response pathways and when selecting/deriving guidelines for impact assessment and management (e.g. assessing the importance of dredging-related light reduction; sediment deposition).

### 3.3.3 Selecting sponge health bioindicators

The usefulness and purposes of assessing bioindicators during pre-development surveys has been discussed previously for corals and seagrass (see sections 3.1.3 and 3.2.3), and these points are equally relevant to sponges. Based on laboratory experiments, bleaching and necrosis are relevant visual indicators of sub-lethal stress in sponges.

- Recording the extent of any bleaching, necrosis and sediment accumulations can be incorporated into pre-dredging surveys to establish the baseline health of sponge populations, and would help differentiate impacts associated with dredging activity from pre-existing sub-optimal (but natural) conditions.
- Baseline surveys should be performed at appropriate temporal and spatial scales to allow the assessment of natural disturbance events on sponge health over different seasons.

### 3.3.4 Assessing recovery potential and pathways

Sponges utilise a number of reproductive strategies, including asexual (e.g. budding, fragmentation) and sexual (e.g. oviparity, viviparity, gonochorism, hermaphrodism) processes (Abdo et al. 2008). The timing of sexual reproduction and the habitat requirements for larval settlement are not clear. As such there is very little information to assist in determining recovery potential and the pathways by which recovery could occur.

# 4 Monitoring and management

## 4.1 Monitoring site selection

Section 3.6.2 of the main body of this technical guidance described the purpose of establishing monitoring sites for informing management and for assessing compliance with approval conditions. It also discusses the purpose of reference sites which are important for providing an evidence base to interpret impact monitoring site data and help disentangle project-attributable impacts from other (natural or human-induced) impacts at local and regional scales.

Impact monitoring sites should be established after the zones of High Impact, Moderate Impact and Influence have been identified for both the ‘management targets’ (i.e. the likely best case scenario based on less conservative, *probable* impact thresholds) and ‘EPOs’ (i.e. compliance limits derived from the likely worst case scenario, which are based on more conservative, *possible* impact thresholds) consistent with the guidance set out in the main body of this technical guidance. The purposes of monitoring in the two sets of zones will differ, and they are considered separately in sections 4.1.2 and 4.1.3. Reference sites on the other hand are applicable to both sets of zones and the key considerations and advice on selecting these sites is provided in section 4.1.1 below.

### 4.1.1 Reference sites

These sites are critically important for interpreting data collected from monitoring sites for informing management, and for assessing compliance with approval conditions.

True reference sites should be unaffected by the activity in question. However, given the potential influence of dredge generated suspended plumes can extend 10s of kilometres from the source under some circumstances, sites that meet these requirements may be at significant distances from the impact monitoring sites.

- Although there will never be identical reference sites for every impact site, best endeavours should be made to match the biological and physical characteristics of the monitoring and reference sites as far as practicable.
- Benthic community composition will never be identical to that at impact monitoring sites, but effort

should be made to find suitable sites with similar species composition and cover, noting that the characteristics of the biological community will be influenced by the physical characteristics of the site such as substrate type, depth, water clarity and exposure to energy.

- There are often strong inshore-offshore gradients in these physical characteristics and hence distance offshore is likely to be an important consideration. Particular attention should be paid to the proximity to, and potential influence of, riverine discharges to locate sites to avoid confounding interpretation if a river-flow event(s) occurs during dredging

It should be noted that the EPA may accept the establishment of 'operational' reference sites within the predicted zone of influence (see section 3.6.2 of the main body of this technical guidance). It is generally easier to find similar sites to those being monitored and the closer proximity helps logistically, but these sites do not replace the need for true (remote) reference sites that remain unaffected by dredging.

- It is important to recognise that the 'operational' reference sites can be affected at some time by dredging and if so they may become invalid for compliance purposes.
- Remote reference sites are those that are unaffected by the dredging activity and are required to assess compliance with environmental protection outcomes (EPOs). The term remote is used here because these sites are generally located at significant distances from the management and compliance monitoring sites but the overarching requirement is that they are unaffected by the activity being regulated.
- Reference monitoring sites for seagrasses would ideally be located within 20 km of sites potentially affected by dredging, have comparable seagrass assemblages and cover, and have physical characteristics as similar as possible to the dredge impact monitoring sites in terms of wind speed and fetch distances, prevailing wind direction, exposure to currents and waves, sediment type and water depth. Most of the variance in seagrass cover was driven by differences among locations separated by tens of kilometres, and these differences corresponded with natural patterns in light intensity, in turn a function of wind speed and direction (McMahon et al 2017b).

### 4.1.2 Management sites

The purpose of these sites is to provide data to inform adaptive management to meet the management target and in doing so, achieve a better outcome than the predicted most likely worst case and hence ensure that prescribed limits (in EPOs) are not exceeded. Sites should be established relative to the location of the zones for management, with a particular focus on the ZOMI (where some level of reversible impact might be acceptable) and near the boundary of the ZOMI and the ZOI (where there can be some pressure but no resultant impacts to benthic communities).

Although not required for the management of dredging programs, it would be useful to establish some of these sites within the predicted ZOHI (i.e. where impacts and effects are likely) and strategies established to record both pressure (type(s) and their associated intensity, frequency and duration) and the responses of the resident benthic communities to those types and levels of pressure. The information generated will be extremely useful for understanding the actual responses of local biota to measured levels of pressure, and assist in refining management triggers and strategies during implementation of the current project, and to inform the assessment and management of future projects (see section 3.6 of the main body of this technical guidance).

The key pressure parameters in both the near-field and far-field are turbidity (as a measure of TSS) and PAR availability at the seabed, whereas sediment deposition becomes increasingly important in and adjacent to the near-field. Issues to consider include:

- PAR and turbidity sensors are commonly used with telemetry systems to provide near-real time data to inform management. If TSS concentrations are measured concurrently (e.g. gravimetrically), these data can be used with the PAR (or derived LAC) and turbidity data to establish predictive relationships between these three variables. These data should be collected as a priority and should also be collected over time to assess natural patterns in the TSS:NTU:PAR/LAC relationships and to determine if and how they change during dredging (see section 2.1.1).

- Sediment deposition is more difficult to measure. Sediment traps provide total gross sediment accumulated over the deployment period (i.e. typically days to weeks). More recently, sensors and data logging systems have been developed through the DSN to provide a proxy for 'net' sediment deposition rates on coral morphologies that are susceptible to sediment smothering (i.e. massive and encrusting). These sensors can provide information on sediment deposition rates over short (i.e. minutes to hours) timescales.
- It is recommended that some redundancy be built into pressure monitoring programs to overcome or ameliorate the implications of instrument failure.

The responses to pressure will vary according to the benthic community type and, for a given pressure, will be greatest in the taxa that are most susceptible to that pressure. For those reasons, and where practical, it is strongly recommended to establish pressure monitoring sites at locations (or at locations immediately upstream from the source of pressure) that have the taxa (or functional forms) for which specific tolerance thresholds have been defined and were used in the impact prediction. When coupled with biological response monitoring (e.g. using bioindicators), this approach will generate validation data to inform adaptive management (primary purpose) but importantly it can assist in verifying impact predictions and refining/revising response thresholds (and management triggers) to local conditions.

- In the case of corals, if key species are not present, or the sites where they occur are otherwise unsuitable, then try and identify sites where coral morphology matches the functional form of the relevant species used to support impact predictions (e.g. foliose, massive, branching).
- Similarly, in the case of sponges, try and identify sites with species used to support the impact prediction or with similar functional forms (e.g. cup) and modes of nutrition (e.g. phototrophic).
- Knowledge of the history and extent of natural mortality of benthic taxa is useful information for management purposes. The extent of any recent mortality of seagrass, and to a lesser extent sponges, is not readily identifiable due to rapid decomposition and the lack of a residual skeleton. However, corals leave a carbonate skeleton making it possible to visually assess recent mortality and the extent of any recent coral mortality should be recorded prior to and during implementation.
- Many of the indicators of sub-lethal stress in corals and sponges (see sections 3.1.3 and 3.3.3) are readily identifiable in the field and should also be recorded. Particular attention should be paid to incorporating these indicators as variables to monitor prior to (i.e. to establish a non-impacted state) and during dredging, and to establish the monitoring sites accordingly.

### 4.1.3 Compliance sites

The selection of compliance monitoring sites should be considered separately from those used for management as they are explicitly linked to the zones established through the relevant conditions of approval (i.e. based on the EPO boundaries described previously - not the zones used for management). The parameters to be measured to demonstrate and assess compliance are typically specified by the Ministerial Conditions of approval (e.g. no change in coral cover; maximum permissible change in coral cover) and may not require monitoring until dredging activities have ceased. In some instances the locations where those assessments are to be undertaken may also be prescribed (e.g. designated reefs).

However, in the absence of specific requirements, and to provide confidence in assessments of compliance, sites for assessing compliance for a particular zone should be located as close as practicable to the boundary of the zone that is closest to the source of suspended sediment generation and hence more likely to experience effects than sites that are more distant from the source of pressure (see Figure 2 of the main body of this technical guidance). If no unacceptable effects are detected at these locations, it is reasonable for the proponent to infer, and the regulator to accept, that there would be a low likelihood of any unacceptable effects in areas of that zone that are further from the source of pressure. Furthermore, it is prudent to monitor dredging-related pressure at these sites as well, even if not prescribed in Ministerial Conditions. These data can be used during dredging to provide confidence that pressures are indeed lower than at equivalent monitoring sites (which are located closer to the dredging and should have higher intensity and duration of pressure), or to trigger investigations against the EPO if levels of pressure become of concern. The data can also provide an additional line of evidence for interpreting and reporting against the EPO for compliance reporting/auditing purposes after dredging is completed. As such:

- The ZOMI EPO monitoring sites should be on relevant benthic communities adjacent to the ZOHI boundary.
- The ZoI EPO monitoring sites used to demonstrate no detectible effect on benthic communities should be adjacent to the outer boundary of the ZOMI.
- Compliance monitoring for the outer boundary of the ZOI is not generally required but there may be requirements to document the realised ZOI. Typically, this is undertaken by mapping plume extent over the duration of the dredging program using aerial photographic imagery or remote sensing imagery (e.g. MODIS; Terra AQUA) using algorithms to convert reflectance signals to turbidity, TSS and PAR availability at the seabed.
- It is useful to monitor relevant pressure parameters concurrently to help infer the cause of any recorded impacts to benthic communities (i.e. provide multiple lines of evidence to support decision-making; see section 4.3).

## 4.2 Selecting monitoring indicators and trigger values

It is not uncommon to use a three-tiered management regime comprised of primary, secondary and tertiary indicators (see section 3.6.3 of the main body of this technical guidance) to assess environmental status against the objectives and trigger management as required. However, the number of levels within the adaptive management system, the indicators and the numerical values selected for use at each level are discretionary, and will depend on factors such as the confidence surrounding: 1. guideline values; 2. predictions; and 3. importance of the biological communities at risk.

Management will generally focus on the Zone of Moderate Impact, with particular attention given to the management triggers associated with the outer boundary of the Zone of Moderate Impact (i.e. between the ZOMI and the ZOI) to ensure no detectible effects of dredging on the benthic communities in the ZOI. However for benthic communities that are unlikely to recover from loss or serious damage within a reasonable (5 years) timeframe (eg. tropical seagrass habitats with no seedbank; temperate Posidonia spp. meadows), management may also need to focus on the inner boundary of the ZOMI.

For the purposes of efficiency and ensuring management is undertaken in a timely manner, most triggers, particularly at lower tiers, will be based on measurements of pressure. Greater confidence in decision making can be achieved by utilising direct measures of biological health - but the tradeoff is timeliness. Many adaptive management programs incorporate a combination of physical and biological indicators, particularly at the higher tiers and where the biological communities at risk are particularly sensitive/important. Considerations regarding the use of pressure-based triggers, and biological-based triggers are discussed in sections 4.2.1 and 4.2.2 respectively.

### 4.2.1 Pressure-based indicators

It is recommended that pressure-based management triggers be derived from the guidelines used for impact prediction such as those presented in Appendix A for corals, seagrass and sponges. These guidelines can be used to derive and establish early warning triggers, based on pressure, to alert dredge operators before management targets are likely to be breached. This approach would provide a clear link between the identified pressure-response pathways used to predict impacts and the actual responses of the biota to those pressures. When used in combination with measures of benthic community response, the approach generates data to 'validate' the predictions and opportunities to fine-tune management based on the comparisons between predicted and actual responses of the local biota to a given level of dredging-related pressure.

In most cases the guidelines used to predict impacts will be comprised of a numerical value for a pressure unit(s) and an averaging period (i.e. a temporal component). It has been established that the most relevant indicators of dredging-related pressure outside of the near-field are turbidity (measured as NTU) and associated light availability at the seabed (measured as PAR at the seabed) and reported as DLI. These parameters are readily and routinely monitored in the field and can be telemetered for use in near real time.

Using these indicators for the purposes of example, a simple approach to apply safety factors would be to set higher average daily light levels and/or lower turbidity, and/or a shorter integrating duration as alert and early warning triggers than the guidelines used to predict the boundaries between zones. Meeting the guidelines would provide confidence that the existing management regime is adequate. Breaching the guidelines would alert dredging managers that conditions are deteriorating and provide the opportunity to adapt management accordingly.

Examples of where these pressure-based indicators from Appendix A have been modified to provide three-tiered management systems for coral, seagrass and sponge communities at the boundary between the zones of Moderate Impact and Influence are set out below. In these examples the tier one trigger is an alert value, the tier two trigger is a warning value and the tier three trigger is equivalent to the management target. In addition, advice is provided on using guidelines derived from baseline conditions that could also be used to derive management triggers for corals in particular, noting that many of the considerations (but not necessarily the guideline values) are also relevant to other BCH taxa.

Similar considerations apply to establishing indicators and triggers for the boundary between the zones of High Impact and Moderate Impact but they are less straight forward to develop and select because significant levels of impact may be acceptable but on the proviso that full recovery occurs within 5 years.

The indicators and numerical values that are ultimately chosen will depend on factors such as the confidence surrounding the predictions and the significance of the biological communities at risk.

*Coral*

Examples of management triggers for use at the boundary between the zones of Influence and Moderate Impact for corals are presented in Table C2. These values were derived from the guidelines in Table A2. In this example the same numerical values as the '**possible-effects**' guidelines are used for the tier 1 triggers, but integrated over 7 days instead of 14. The tier 2 trigger values use the same numerical values as the '**probable-effects**' guidelines but are integrated over 7 days instead of 14. The tier 3 trigger values are identical to the '**probable-effects**' guidelines. Note there is no specific basis for the numerical values shown in Table C2, they were arbitrarily selected for the purpose of example.

**Table C2: Example of interim targets (tiered triggers) that could be used for corals as part of a tiered management system designed to provide early warning of conditions approaching the management target for the boundary between the zones of Moderate Impact and Influence (from Table A2)**

| | Threshold type | Averaging period | MT | | |
|---|---|---|---|---|---|
| | | | NTU | SSC | DLI |
| Tier 1 trigger | Running mean (days) | 7d | >6.5 | >11.7 | <2.3 |
| Tier 2 trigger | | 7d | >10.0 | >18.0 | <1.1 |
| Tier 3 trigger | | 14 d | >10.0 | >18.0 | <1.1 |

Note: Suspended sediment concentration (SSC) units are mg $L^{-1}$ and daily light integral (DLI) units are mol photons $m^{-2}$ $d^{-1}$. NTU is nephelometric turbidity units. SSC was calculated from NTU where SSC= NTU x 1.8.

*Seagrass*

Management triggers for the ZOMI/ZOI boundary can be derived for seagrass using the guidelines in Tables A5 and A6 by applying the same logic as described for corals. These guidelines are based on the key pressure-response pathway (i.e. light stress) for seagrass, and data for assessment against the triggers can be obtained relatively simply in real time using appropriate sensors and telemetry systems. The guidelines include a combination of numerical values for average daily light (DLI) and for the duration of low light periods within a 2-week period. In this case early warning triggers could be derived from the guidelines by setting higher average daily light levels than the management target; higher permissible low light intensity and/or a shorter duration.

An example of where these aspects have been modified to provide a three-tiered management system for a mixed meadow is provided in Table C3. In this example the tier 1 triggers are derived from the **possible-effects** guidelines for mixed meadows in Table A5. The first duration (applied to every 2-week averaging period for the duration) was halved to 6 weeks but the mean DLI remained the same. The second duration (days) was not altered but the mean daily DLI value was raised by 1.5x. The tier 2 triggers were also derived from the same **possible-effects** guidelines in Table A5, the only difference being the second mean DLI was raised by 1.25x. The tier 3 triggers are identical to the **probable-effects** guidelines in Table A6. Note there is no specific basis for the numerical values shown in Table C3, they were arbitrarily selected for the purpose of example.

**Table C3: Example of interim targets (tiered triggers) that could be used for seagrass as part of a tiered management system designed to provide early warning of conditions approaching the management target at the boundary between the zones of Moderate Impact and Influence**

| | Two-week averaging period over duration | | Within a two-week averaging period | |
|---|---|---|---|---|
| | Duration (weeks) | Mean DLI* (for each two-week period) | Duration (days) | Mean DLI* (Daily) |
| **Mixed Meadow** (based on total biomass of all species in a multi-species meadow) | | | | |
| Tier 1 trigger | >6 | <13.1 | >5 | <3 |
| Tier 2 trigger | >12 | <13.1 | >5 | <2.5 |
| Tier 3 trigger (also Management Target) | >12 | <8.9 | >10 | <2 |
| | >6 | <5 | >10 | <2 |

*DLI = daily light integral as mol photons $m^{-2}$ $d^{-1}$

*Sponges*

Sponges can be susceptible to suspended and deposited sediments directly and, in the case of phototrophic taxa, indirectly through increased turbidity and reduced benthic light (see section 3 of Appendix A).

For the purposes of example, management triggers were derived from Table A9 using similar approaches to those used for corals and seagrasses above. The management triggers are designed to be applied at the ZOMI/ZOI boundary to protect sponge communities containing phototrophs and are presented in Table C4.

The tier 1 trigger values are derived from the **possible-effects** guidelines in Table A9, by raising the long-term DLI value by 1.5x and reducing the duration of the medium-term integration period to 5 days. The tier 2 trigger values were also based on the same **possible-effects** guidelines, except the durations of the long-term and medium-term integration periods were reduced to 14 days and 5 days respectively and the short-term mean DLI value increased to 0.5 instead of 0.1. The tier 3 trigger values are identical to the probable-effects guidelines in Table A6. There is no specific basis for the numerical values shown in italics, they were arbitrarily selected for the purpose of example.

**Table C4: Example of interim targets (tiered triggers) that could be used for sponges as part of a tiered management system designed to provide early warning of conditions approaching the management target boundary between the ZOMI and ZOI. In this case light availability at the seabed is the key indicator of dredging-related pressure.**

| | Long term | | Medium term | | Short term | |
|---|---|---|---|---|---|---|
| Trigger level | Mean DLI** | duration (days) | Mean DLI** | duration (days) | Mean DLI** | duration (days) |
| Tier 1 trigger | <4.6 | 28 | ≥1 - 3.1 | 5 | <1.0 | 1 |
| Tier 2 trigger | <3.1 | 14 | <1.0 | 5* | <0.5 | 1 |
| Tier 3 trigger | <1.0 | 28 | <0.1 | 7* | n/a | n/a |

* if *Carteriospongia foliascens* is present (or other sensitive phototrophs) then a duration of ≤2 days may be more appropriate

** DLI = daily light integral as mol photons $m^{-2}$ $d^{-1}$

*Triggers based on background conditions*

Although it is recommended that management triggers are based to some degree on the guidelines used to underpin the impact assessment (and hence provide location specific validation data for adaptive management and improved confidence for future projects) there are some alternative approaches that could be considered, particularly for developing the lower tiered (early warning) management triggers or simply to provide confidence. These are outlined below and although the numerical values are specific to corals, similar considerations would apply when adopting these approaches for other taxa.

Using percentiles of background (ANZWQG (2018) approach)

Guidelines calculated from the Barrow Island water quality data set using the ANZWQG (2018) $P_{50}$–$P_{80/20}$ approach were compared with the conservative 'possible effect' thresholds (see Table A2) derived using the same water quality data set and the $P_{50}$–$P_{80}$ approach was found to be far more conservative (see Fisher and Jones 2018).

- The $P_{80/20}$ values calculated from good quality background data sets, that do not include the effects of extremely rare high intensity events (e.g. cyclones), are very likely to represent 'safe' conditions for corals growing in 'normal' conditions (e.g. not at their local extreme depth limit). This is not to suggest that above these levels is 'un-safe', rather there is high confidence that if the median value ($P_{50}$) of monitoring data remains below the $P_{80}$ (or above the $P_{20}$ for light) of baseline (pre-dredging) conditions, corals are very unlikely to be affected. This could be used very early in a tiered management system (or even sitting outside the formal management system; to provide re-assurance if met but not trigger formal management if not met).

- $P^{95/05}$ values derived from data sets as described above can also provide a reasonably high confidence that corals growing in 'normal' conditions (see above) are unlikely to be affected.
- Care should be taken when calculating and using higher (or lower) order percentiles ($P_{95}$,$P_{99}$; $P_{05}$,$P_{01}$) as they are strongly influenced by extremes in the data sets used to derive them. Confidence exponentially decreases the further percentiles are from the median value. Seasonal factors may also need to be taken into consideration (see ANZWQG 2018 for advice).

Using multiplier of percentiles of background

A similar comparison was undertaken using Barrow Island water quality data to determine what multiplier of background conditions equated to the 'possible' and 'probable' effects thresholds described previously by Fisher and Jones (2018) and suggested for use to define the outer boundary of the ZOMI (see Table A2). The mean multipliers required to convert the P50 value of baseline turbidity and light stress to the (conservative) possible effects guidelines were 2.2x and 1.4x. The lowest multiplier values for any site for turbidity and light stress were 1.5x and 1.2x respectively.

- Guidelines that are likely to represent 'safe' conditions for corals growing in 'normal' conditions (e.g. not at their local extreme depth limit) can be derived from good quality background data sets, that do not include the effects of extremely rare, high intensity events (e.g. cyclones), by applying a factor of 1.5 and 1.2 for turbidity and light stress respectively. As with the percentile-based guidelines, this is not to suggest that breaching these levels is 'un-safe', rather that corals are very unlikely to be affected at these levels.
- Factors of 2.2 and 1.4 could be applied to turbidity and light stress respectively (as above) but, given spatial variability and other factors, there is less confidence that corals growing in 'normal' conditions (see above) are unlikely to be affected at these levels of pressure.

### 4.2.2 Biological indicators

Direct measures of benthic community health (i.e. bioindicators) provide more confidence that the benthic community is being adequately protected than can be inferred from measurements of pressure alone. When considering the most appropriate bioindicators to incorporate, it is important to consider the pressure-response pathways that are likely to operate and choose bioindicators accordingly. Furthermore, the level of confidence in these assessments can be improved if the relevant bioindicators assessments are targeted to the most susceptible taxa within that benthic community (i.e. use of sentinel species).

The trade-off for the increased confidence associated with using direct measures of health for decision making relates to the level of difficulty that is typically associated with obtaining measurement data for bioindicators in the field, and the time required to analyse and interpret the results, which may delay management intervention and potentially lead to exceedance of EPOs.

A range of bioindicators to different dredging pressure response pathways, and that may prove useful for managing impacts to key tropical marine benthic community types, are outlined below.

*Coral*

A number of sub-lethal bioindicators of coral health have been identified that can be linked to one or more types of dredging-related pressures. Furthermore, taxa that are particularly susceptible to those dredging pressures are identified and could be used as sentinels of stress on the coral community as a whole. These are outlined below according to the pressure:

Regional pressures

Coral bleaching reduces the ability for corals to obtain energy from photosynthetic pathways (via the algal symbionts). DSN research has also shown that bleaching significantly reduces the ability of corals to actively remove sediments from their surfaces (Bessel-Browne et al 2017a). This loss of sediment rejection capability makes them much more vulnerable to sediment smothering if dredging happens to coincide with warm-water bleaching events or where corals have bleached for other reasons (e.g. dredging related effects covered previously).

- Coral discoloration or bleaching is a useful sub-lethal bio-indicator that may signal that additional management measures need to be put in place to reduce dredging related pressure to below that which would be normally tolerated by healthy (un-bleached) corals.

Sediment accumulation

Sediment deposition and sediment smothering of coral is arguably one of the most significant, but least understood, pressure-response pathways resulting in mortality of some types of corals during dredging programs (Jones et al. 2016). When sediment deposition rates exceed the rate of the corals' ability to self-clean, or their energy (lipid) reserves are depleted from combatting repeated or chronic deposition events, sediments will remain on the surface and begin to accumulate over successive days. Smothered corals can bleach within a few days and pockets of sediment can lead to lesion formation and tissue mortality.

- The foliose (e.g. *Montipora* spp.) and massive species (e.g. *Porities* spp.) are relatively easy to identify in the field and are far more sensitive to deposition than the branching species. Individual colonies of these species could be usefully monitored over time as sentinels of the level of sediment deposition pressure, particularly in areas of very high SSC that can occur within a few kilometers of large-scale dredging operations.

A useful sub-lethal bio-indicator of sediment deposition stress identified through DSN research is mucus sheet production in massive *Porites* spp. corals. The results of the studies showed a close association between mucous sheet formation in *Porites* spp. colonies and sediment load from dredging (Bessell-Browne et al. 2017b).

- The % coverage of the surface of *Porites* spp. colonies by mucus sheets and the number of sheets produced over time are useful bioindicators of exposure to sediments, particularly when compared to pre-dredging and/or reference sites. In the absence of location specific information, greater than ~2% prevalence of mucus sheets may indicate sediment deposition stress.

Light reduction and elevated suspended sediments

The most sensitive species to these stressors (from laboratory experiments) were the branching species *Pocillopora damicornis* > *Acropra millepora* followed by the massive *Porites* spp.

- *P. damicornis* and *A. millepora* (or other Acroporids with similar morphologies) are relatively easy to identify in the field and their health status (e.g. degree of bleaching, partial mortality) could be usefully monitored over time as sentinels of the overall health of the coral community they are part of.

*Seagrass*

Direct measures of seagrass health provide more confidence than can be inferred from pressure-based indicators alone, but typically they are more difficult to implement and the feedback for management is sometimes delayed whilst data are being analysed and interpreted. The DSN research identified four robust bioindicators of light-reduction stress, and one potential indicator of sub-lethal burial stress that is valid under some conditions:

Light stress

- Maximum electron transport rate ($ETR_{MAX}$) is a measure of photosynthesis and is very early in the pressure-reponse pathway for turbidity-mediated light stress. $ETR_{max}$ will be most be useful in situations where short-term changes in seagrass condition need to be monitored and assessed, or to define the Zone of Influence of a sediment plume over relatively short time periods and where the protection of seagrass from any adverse effects is a management objective. It is noted that $ETR_{max}$ is relatively complicated to measure and applying this indicator in an uncontrolled field setting is not very practicable.
- Carbohydrate concentrations in the rhizome is useful for assessing changes in light climate over a longer-period than $ETR_{MAX}$. Concurrent measurements of this indicator at impact and reference sites could be used to determine the boundaries between the zones of Influence and Moderate Impact. Applying this indicator for management purposes could be problematic because of the time required to undertake analyses and trigger action if necessary.
- 'Above-ground biomass' and 'total biomass' are useful and practical indicators for detecting impacts on seagrass from a range of pressures including reduced light, particularly when analyses factor in data from relevant reference sites. As such, the thresholds based on these variables can be used for compliance assessment purposes (e.g. to determine the boundary between the zones of Influence and Moderate Impact). This indicator could also be used for adaptive management but feedback will be delayed due to the time required to undertake analyses.
- Total biomass sampling typically requires divers to collect samples but above-ground biomass could be estimated from calibrated shoot density/cover measurements taken from seabed imagery using diver-less techniques. There are trade-offs involved and although diver-less image analysis techniques can be efficient, the data are likely to be more robust when physical samples are taken and allowing both above-ground biomass and total biomass to be determined.

Vertical rhizome growth

- The extent of vertical rhizome growth will be useful for assessing the extent and history of sediment deposition and seagrass burial at a site and sub-lethal effects on seagrasses over week to month timescales if the site is not subject to very low light levels or organic-rich sediments.

*Sponges*

There are no generic bioindicators that are applicable to all sponges given the high species diversity and range of nutritional modes within this group, however there are some useful bioindicators for phototrophic assemblages in particular. There are also particularly sensitive/conspicuous taxa that could be used as sentinels to provide early warning of pressures on the broader filter feeding community that are part of.

Light stress

- Prolonged light stress due to increased light attenuation can lead to mortality, so early detection of discolouration (bleaching) in phototrophic sponges can indicate that phototrophic sponge populations are under stress and allow management intervention to prevent mortality.
- Discolouration would need to be used carefully as a bioindicator given that discolouration could also be related to natural causes, or be evident as a diel pattern in some species. As such assessments using this indicator should include comparisons with sponges from unaffected reference sites to account for natural changes, and use observations taken at defined times of the day to account for any diel patterns that may be evident (Penida et al 2017).
- This bioindicator would be particularly useful when considered in combination with light availability triggers such as the examples in Table C4.

- Cup and phototrophic species such as *Carteriospongia* spp. are particularly sensitive to changing water quality conditions that affect light transmission and exhibit bleaching in response to reduced light availability. Individuals of these taxa are useful as sentinels of light stress.

Sediment stress

- Sediment stress can cause necrosis and lesion formation (i.e. partial mortality) which may be irreversible in some sponge species and lead to rapid mortality (e.g. in *C. foliascens*).
- Observations of lesion formation and necrosis could also be useful bioindicators through comparisons between monitoring and reference sites.
- Above average mucus sheet production, oscula closure and tissue regression were found in some species subjected to elevated TSS and sediment deposition. These traits could be considered as bioindicators of turbidity-related and deposition-related stress in some sponge species.

## 4.3 Multiple lines of evidence

Management intervention to prevent impacts to benthic communities requires information on variables that respond early in the relevant pressure-response pathways and that are linked to the activity causing the pressure. Bioindicators are direct measures of sub-lethal effects or impacts, and can be used in conjunction with data on pressure fields (e.g. Suspended and deposited sediments, and seabed light) to provide multiple lines of evidence to improve confidence in decision-making. As such it may be prudent to collect data on relevant pressure parameters and more than one bio-indicator for each biological community type wherever feasible, even if formal decision-making thresholds are not based on all of the data sets.

Section 4.2.2 above sets out a number of bioindicators for three key benthic community types found in northern WA. The section on sponges above identifies a bioindicator early in the pressure-response pathway for light-mediated sub-lethal responses (e.g. bleaching) that can be used for early warning; the others linked to the effects of suspended and deposited sediments appear to be further along that pathway and potentially signify un-recoverable impacts. As such they are probably more useful for confirming that impacts have occurred.

Although the bioindicators of stress described in section 4.2.2 above are scientifically based, they are not specific to dredging pressures alone and can only be confidently applied using comparative assessments against measurements of the same bioindicator at reference sites unaffected by the dredging. This is necessary to ensure any measured changes in the bioindicator are related to the dredging and not some regional stress such as a marine heatwave. These comparative assessments, coupled with contemporaneous measurements of pressure at monitoring and reference sites, are key planks of a multiple lines of evidence approach for assessing and managing impacts to all benthic communities for both management and compliance purposes.

Furthermore, if evidence can be provided that the management objectives have been met for a set of zones that are significantly smaller in spatial extent than those that represent the maximum allowable impact (i.e. the EPOs), then it would provide a strong line of evidence to infer that the EPOs have also been met. Under these circumstances, using a risk-based approach it is reasonable to expect that the extent of monitoring required to demonstrate compliance against the EPO would be significantly reduced, and in some instances may be eliminated entirely. It also signals that a better than approved environmental outcome was achieved. These are important considerations and provide further reasons to design the adaptive monitoring and management program to include the relevant indicators of biological responses to pressures that are expected to be used to assess compliance with the EPOs established through the Ministerial Conditions of approval.

Environmental Protection Authority 2021, *Technical Guidance – Environmental impact assessment of marine dredging proposals*, EPA, Western Australia.

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