



## Assessing the impact of sediments from dredging on corals

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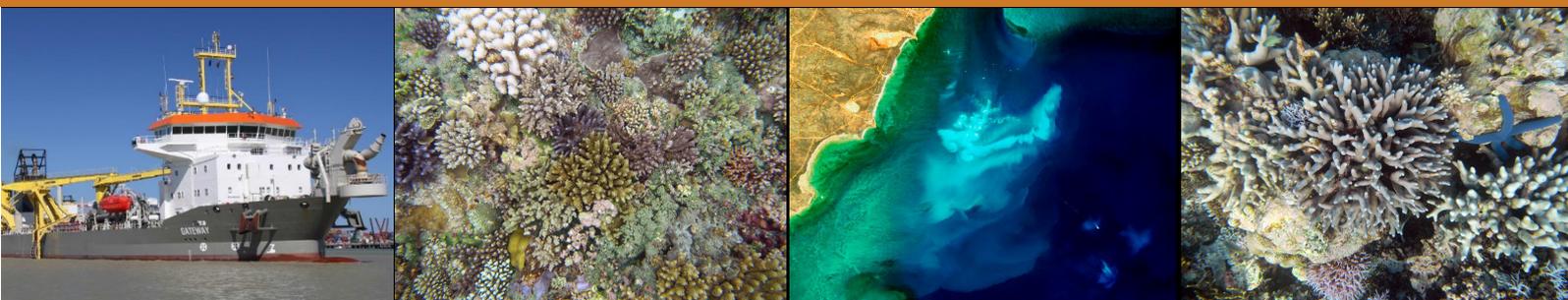
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## WAMSI Dredging Science Node

### Theme 4 Report

Project 4.1

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## WAMSI Dredging Science Node

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

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

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

### Funding and critical data

### Critical data



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## Front cover images (L-R)

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

Image 2: Close up image of the reef flat at Scott Reef (Source: AIMS)

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

Image 4: Close up image of the reef flat at Scott Reef (Source: AIMS)

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## Executive summary

The release of sediment into the water column by dredging and dredging-related activities such as dredge material placement (spoil disposal) increases water cloudiness (turbidity) causing a range of effects on the underlying habitats. There is a pressing need to derive relevant water quality thresholds to better manage dredging projects whilst underway, or for use beforehand at the environmental impact assessment (EIA) stage, to predict the likely environmental outcomes. Increased confidence in outcomes would result in less compliance monitoring, in turn imparting significant cost savings to dredging proponents and more timely approvals<sup>1</sup>.

A comprehensive review<sup>2</sup> was conducted of all available scientific literature on the effects of sediments on corals to guide subsequent laboratory and field-based studies associated with threshold development. To frame the review, a conceptual model was developed using the US Environmental Protection Agency (USEPA) causal/diagnosis decision information system framework (CADDIS). The model was used to identify the likely proximal stressors and potential (biologically plausible) cause-effect pathways, how they are interlinked, and their relationship with interacting factors such as natural turbidity events and seasonal cycles. Potential damage associated with dredging and turbidity-generating activities were then separated into *direct* and *indirect* effects. The *direct* effects included the removal of hard and soft substrate i.e. the dredging footprint, and smothering of the seabed at the dredge material placement sites (spoil disposal grounds). The *indirect* effects were associated with mobilization of sediments into the water column (i.e. turbidity or plume-generation), and formed a second and larger group of cause-effect pathways. The indirect effects were subsequently divided into (1) *chemical* effects (such as the release of FeS-rich sediments and nutrients and legacy contaminants) and (2) *physical* effects. The review is primarily concerned with the *physical* effects.

direct effects (dredge footprint)

indirect effects (plume migration) → (a) chemical effects (release of nutrients, legacy contaminants etc)  
(b) physical effects (light attenuation, sedimentation etc)

The model showed that the key proximal stressors affecting corals are: (1) light attenuation affecting photosynthesis (autotrophy), (2) high suspended sediment concentrations (SSCs) affecting feeding processes (heterotrophy) and, (3) sediment deposition resulting in smothering of corals and restricting solute exchange and light. The mechanism(s) whereby each of these proximal stressors affects corals are discussed. From the model it is clear that the proximal stressors are highly interlinked, and that they could act alone or in combination, and that the most relevant parameter(s) may change according to dredging activities, metocean conditions, distance from the dredge etc.

Surprisingly, water quality conditions during large-scale dredging programs have not been very well described (in the public literature) for the key proximal stressors. To that purpose, an analysis was also conducted of water quality information collected during several major dredging programs<sup>3,4,5</sup> in tropical Western Australia in recent years. In the near-field environment (~100s metres from an operating dredge) sediments released into the water column and moving out of the immediate dredging area were primarily silt-sized (4–62 µm). These measurements are consistent with analyses of the composition of the seabed after the dredging programs described in the review, which show a build-up of predominantly silt-sized particles. Water quality was profoundly affected by dredging, as anticipated; however, a significant and characteristic feature was that even close to dredging activities turbidity events are often very ephemeral. Order of magnitude changes in water

<sup>1</sup> EPA (2016) Technical Guidance: Environmental Impact Assessment of Marine Dredging Proposals. Environmental Protection Authority, Perth, Western Australia.

<sup>2</sup> Jones R, Bessell-Browne P, Fisher R, Klonowski W, Slivkoff M (2016) Assessing the impacts of sediments from dredging on corals. Marine Pollution Bulletin <http://dx.doi.org/10.1016/j.marpolbul.2015.10.049>

<sup>3</sup> Pluto LNG Development, Burrup Peninsula: WA Environmental Protection Authority Bulletin 1259, Ministerial Statement No. 757<sup>3</sup>

<sup>4</sup> Cape Lambert B project: WA Environmental Protection Authority Bulletin 1357, Ministerial Statement No. 840

<sup>5</sup> Gorgon Gas Development Barrow Island Nature Reserve: WA Environmental Protection Authority Bulletin 1221 Ministerial Statement No. 800

quality can occur in the space of hours and days, but are often interspersed with periods of otherwise quite good water quality. This has implications for designing and interpreting information collected from water quality monitoring programs and the setting of thresholds (see below). SSCs over very short term periods (hours) could reach levels of hundreds of  $\text{mg L}^{-1}$  but over days to weeks the worst case scenarios averaged 10s of  $\text{mg L}^{-1}$  and over a period of a month averaged  $<10 \text{ mg L}^{-1}$ .

Scattering and light attenuation by suspended sediments occurred rapidly in the water column, with the shallow reef environment routinely experiencing semi-dark, caliginous or diurnal 'twilight' periods. Complete loss of light at moderately elevated SSCs occurred quite regularly, especially at deeper sites. Hyper-spectral analyses of light quality underneath a plume, which are some of the first measurements of their kind for dredging in reefal environments, showed different wavelengths of light are preferentially attenuated. There was a more immediate loss of useful blue light and a shift of the underwater spectrum to less photosynthetically useable yellow-green light. This is significant because photosynthetically active radiation (PAR) which is frequently monitored during dredging projects, may be present but not in a very useable form for primary production.

All available laboratory and field based studies of the effects of suspended sediments and sediment deposition on corals were then compiled and reviewed. The experimental conditions used in these tests were also closely scrutinized. This body of literature, which has been generated over the last 30 years, has been reviewed many times, and has shaped our present day understanding of the effects of sediments on corals. The experimental/laboratory conditions in these studies have been highly varied in terms of sediments used, particle size distributions, lighting conditions, test durations, exposure regimes and sediment loading rates. The experimental conditions used were also found to differ quite significantly from the *in situ* conditions described previously. Several studies had used proxies for sediment including carborundum. Many studies, especially those associated with sediment deposition, have also used sands (62–2000  $\mu\text{m}$ ) as opposed to the fine silts and clays which typify a dredge plume. Experiments exposing corals to elevated suspended sediments have also typically been conducted in small aquaria, and not all studies compensated for the fact that there would only be a small amount of light attenuation in the few centimetres of water above the corals in a shallow tank. For example the light attenuation associated with a high 30  $\text{mg L}^{-1}$  SSC may be  $\sim 10\%$  in a shallow aquarium, whereas *in situ*, on the reef and at 5 m, an equivalent SSC would attenuate  $>99\%$  of all PAR. Re-interpreting some of these studies within this review, it is suggested that elevated SSCs *per se* (as opposed to the associated light reduction and sediment deposition) is perhaps not as significant a stressor as previously considered. None of the studies had compensated for spectral changes in light discussed previously, and overall results from these studies are therefore difficult to evaluate.

One of the most significant issues identified by the review was the problem of understanding the effects of high sedimentation caused by dredging on corals. During natural high turbidity events produced by high wind/wave, wave orbital velocities are usually sufficient to keep some of the sediments in suspension. Sediment deposition subsequently occurs (after a settling lag) during quiescent periods and after further entrainment and dilution occurs. In contrast, dredging can result in high SSCs in a low energy water column, where sediments cannot be supported by ambient hydrodynamics. Under such conditions the sediment rapidly falls out of suspension and probably exceeds levels that corals are ever likely to have experienced naturally (unless under extreme circumstances). Once the inherent self-cleaning capacity of a coral has been exceeded, sediments will begin to accumulate on its surface, progressively smothering it in a thick sediment deposit. The ultimate fate of the underlying coral tissue is then localised mortality and lesion formation unless sediments can be resuspended by strong water flow such as during a storm. Understanding the relationship between sedimentation rates and coral health is very important for determining the severity of impact around dredging operations. The problem is that there have been few ecologically-relevant and reliable measurements made of sedimentation over appropriate scales (i.e.  $\text{mg cm}^{-2} \text{ d}^{-1}$ ) because of a lack of suitable instrumentation.

In terms of impact prediction, the review concludes that the proximal stressors associated with dredging (elevated SSCs, light reduction and sedimentation) are very interlinked, that they could act alone or in combination, and that the most relevant parameter(s) may change according to dredging activities, metocean conditions, distance from the dredge etc. This 'protean' nature of suspended sediments i.e. the ability of the stressor parameters to change shape and form, makes it particularly challenging to (a) identify which is the most

relevant or important parameter(s) at any given time (the cause-effect pathway(s)), and (b) establish dose-response relationships. This potentially confounds and confuses laboratory and field experiments and observations and their interpretation. The problem is further compounded by the lack of suitable instrumentation for estimating sediment deposition and consequently a lack of pressure field characterization for one of the key-cause effect pathways.

## Considerations for predicting and managing the impact of dredging

### Pre-development Surveys

The review contained an initial exploration of the temporal and spatial patterns in water quality (SSCs and light attenuation) and an analysis of water column and seabed sediment particle size distributions that can occur during large-scale dredging campaigns. **The information proved invaluable for characterizing the pressure field and for contextualizing the results of past, predominantly laboratory studies.** The significance of this data as a reference point for contextualizing any effects on water quality in future dredging projects is discussed in several other reports, see <sup>6,7,8</sup>. **Dredging proponents should be encouraged to make available for future analysis all data (and metadata) from water quality investigations (i.e. turbidity, surface and benthic PAR, temperature, sediment particle size distributions in the water column and seabed, and associated ecological information) collected during baseline periods and during and after dredging.**

The increased silt content of the seabed from sediments spilled during the dredging process described in the review may have longer term implications for the environment, as the sediments are more easily re-suspended by storms and wind or wave events (see also<sup>7</sup>). Chronic low level turbidity may influence coral recruitment and may result in population and community level effects as opposed to effects on adult colonies. Efforts should be made to establish how long after dredging projects the seabed particle size distributions are skewed towards finer particles, and whether this is having a measureable effect on water quality, and over what time periods. **Studies are needed to understand the longer term legacy effects of dredging i.e. how long after dredging projects the seabed particle size distributions are skewed towards finer particles, whether this is having a measureable effect on water quality, and over what time periods. This necessitates establishing pre-development information on seabed particle size distribution and temporal/spatial variability in water quality.**

### Impact prediction

**The initial exploration of the temporal and spatial patterns in water quality during dredging showed that it is highly ephemeral and to fully capture this, thresholds for impact prediction should be developed at multiple time frames, including both short term, acute periods from hours to days to longer chronic thresholds from weeks to months.** A generally accepted model for how corals tolerate turbidity is that they survive short term periods of high SSCs by shifting between phototrophic and heterotrophic dependence, by relying on energy reserves, and by rapidly replenishing reserves in periods between turbidity events. The ephemeral nature of plumes and the potential for corals to recover from individual turbidity events, means dredging programs can be managed by considering cumulative pressure. Implicit in this concept is that natural turbidity events (or periods of low light), are an integral component of the total pressure. That is, **corals or other epi-benthic organisms cannot differentiate between natural turbidity events and dredging-related events, and they should not be distinguished between during water quality monitoring programs associated with dredging campaigns. Thresholds for impact prediction therefore should be developed over multiple time frames which include**

<sup>6</sup> Fisher R, Stark C, Ridd P, Jones R (2015) Spatial patterns in water quality changes during dredging in tropical environments. PLoS ONE 10:e0143309.

<sup>7</sup> Jones R, Fisher R, Stark C, Ridd P (2015) Temporal patterns in water quality from dredging in tropical environments. PLoS ONE 10(10): e0137112. doi:10.1371/journal.pone.0137112

<sup>8</sup> Abdul Wahab MA, Fromont J, Gomez O, Fisher R, Jones R (2017) Effects of dredging on benthic filter feeder communities, field study. Report of Theme 6 - Project 6.3, prepared for the Dredging Science Node, Western Australian Marine Science Institution, Perth, Western Australia.

### natural turbidity events.

Many environmental impact assessments for large scale dredging projects contain meta-analyses and tabulations of results from many published laboratory-based studies on the effects of sediments on corals for (or to support) impact prediction. This is a logical step, **but caution needs to be exercised in using the information (for that purpose), as not all studies may be suitable. This relates to problems with the types and particle sizes of sediments used, and the use of sediment proxies, as well as problems associated with extrapolating results from laboratory based experiments which have not adequately compensated for light reduction.**

The change in spectral quality of light by dredging plumes identified during the review also has implications for determining minimum light requirements for corals from laboratory-based studies. Lab-based studies typically reduce light levels by using neutral density filters, or by adjusting the intensity or distance of lights above the corals. This will reduce the light quantity but the spectrum will not be notably changed. This could lead to an underestimation of the light required *in situ*. In the field, the symbiotic algae in corals may be able to adjust their photosynthetic and accessory pigments to harvest light from the less optimal wavelengths, but given the highly ephemeral nature of plumes, they may not be able to undergo such photoacclimatory changes over periods of hours or days. **The implications of this are that laboratory based studies could underestimate the amount of light required by phototrophs *in situ*. The potential significance of this should be examined further under controlled laboratory based conditions.**

For impact prediction purposes a much more significant problem than the relevance of experimental conditions from past experiments is a broader issue associated with the existence of the many different and often intertwined cause-effect pathways. **Whilst the laboratory based experiments are useful for understanding physiological responses for specific cause-effect pathways, the information may not be directly suitable for impact prediction purposes unless the consequences of the exposure scenarios are known for all other possible pathways.** For example, laboratory-based studies where corals have survived exposures to suspended sediment concentrations in the high tens of  $\text{mg L}^{-1}$  for several days may not be useful for impact prediction purposes *in situ* if it occurs during calm sea conditions which would result in very high deposition rates (and subsequent sediment smothering). This could lead to much more rapid mortality of corals — and before any effects of the elevated SSCs and light reduction in the water column could occur. **This indicates the need for multiple thresholds for different cause-effect pathways and the use of realistic worst case scenarios to avoid failing to predict an effect when one might occur. The immediate problem, and as emphasized in the review, is that until suitable ways can be developed to measure ecologically-relevant sediment deposition rates *in situ* at appropriate scales (i.e.  $\text{mg cm}^2 \text{day}^{-1}$ ), and relationships to coral health determined, there will always be a great deal of uncertainty associated with impact prediction for dredging near coral reefs. How to deal with this uncertainty is discussed in a subsequent section regarding the water quality threshold development based on the laboratory and field studies.**

One approach to circumventing these problems (the existence of multiple cause-effect pathways<sup>9</sup> and the lack of instrumentation to measure pressure associated with one of them) is to concentrate on the first step in the chain of the interlinked cause-effect pathways, which was identified by the CADDIS model as elevated turbidity. The underlying logic is that elevated turbidity must be indicative of elevated SSCs and light attenuation (by definition), and there may be elevated sediment deposition depending on the sea conditions. In theory a range of turbidity values can be determined, over different time scales, which have, or have not, resulted in environmental effects in past dredging campaigns. The information might have use for impact prediction purposes. The advantages of this approach is that impact prediction is based on a single parameter, that turbidity can be measured relatively easily by *in situ* instrumentation, and that there is information from combined water quality and coral health monitoring programs during baseline periods and dredging phases for several programs to examine the relationship. The disadvantage of the approach is that turbidity is simply being used as a proxy

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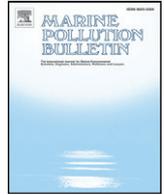
<sup>9</sup> (1) Light attenuation and a reduction in coral phototrophy, (2) high suspended sediment concentrations and a reduction in coral heterotrophy, (3) high sedimentation rates and an increase in sediment smothering.

for the 3 different cause-effect pathways<sup>8</sup>, the relationship between turbidity and sedimentation is poorly understood, and any effects observed will be correlative.

## **Residual knowledge gaps**

The review identified several knowledge gaps needed to be addressed to enhance capacity within government and the private sector to predict and manage the environmental impacts of dredging. Since the purpose of the review was to inform the research program in Theme 4, most of the knowledge gaps are being addressed in the individual projects. Thus Project 4.2.3 is focused on understanding the relationship between turbidity and sedimentation and identification of conditions where high turbidity may or may not result in high sediment deposition. Project 4.2.4 concentrates on the development of new instrumentation for measuring sedimentation in situ, and Project 4.4 strives to the cause-effect pathways either alone or in combination, and establishing dose response relationships for sedimentation and coral clearances rates to identify when smothering may occur.





## Review

## Assessing the impacts of sediments from dredging on corals



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## ABSTRACT

There is a need to develop water quality thresholds for dredging near coral reefs that can relate physical pressures to biological responses and define exposure conditions above which effects could occur. Water quality characteristics during dredging have, however, not been well described. Using information from several major dredging projects, we describe sediment particle sizes in the water column/seabed, suspended sediment concentrations at different temporal scales during natural and dredging-related turbidity events, and changes in light quantity/quality underneath plumes. These conditions differ considerably from those used in past laboratory studies of the effects of sediments on corals. The review also discusses other problems associated with using information from past studies for developing thresholds such as the existence of multiple different and inter-connected cause-effect pathways (which can confuse/confound interpretations), the use of sediment proxies, and the reliance on information from sediment traps to justify exposure regimes in sedimentation experiments.

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## 1. Introduction

Dredging and dredging related activities such as dredge material placement (spoil disposal) releases sediment into the water column where it can have a significant effect on nearby benthic communities (Bak, 1978; Brown et al., 1990; Dodge and Vaisnys, 1977; Erfteimeijer et al., 2012; Foster et al., 2010; Rogers, 1990). Consequently, most large-scale dredging projects require environmental impact assessments (EIAs) and active management when underway (EPA, 2011). The environmental and management issues associated with dredging and construction around ports and harbours have recently been reviewed and guidelines produced for monitoring, management, and mitigation (Foster et al., 2010).

There is a critical need to improve the ability to make scientifically sound predictions of the likely extent, severity, and persistence of environmental impacts associated with dredging especially when conducted close to sensitive habitats such as coral reefs. One of the more practical ways is by water quality monitoring, i.e. measuring the key hazards which are capable of having adverse biological effects. Water quality monitoring can be vessel-based or by fixed or rapidly relocatable mobile 'sentinel' in situ platforms, or even autonomously by vehicles remotely controlled from land (Van Lancker and Baeye, 2015). The use of these technologies allows dredging proponents to be quickly alerted to conditions where environmental damage may occur and modify the dredging activities as required. However, to be useful in this way, the water quality data have to be coupled to thresholds that relate the physical pressure(s) to a biological response(s) and define exposure conditions above which effects could occur.

Similar threshold values for chemical contaminants are usually derived from field data and more frequently from controlled, laboratory-based studies (i.e. exposure studies) where test organisms are incubated with toxicant(s)/hazards over short-term (acute) and longer term (chronic) time periods (Chapman, 2002). Many such studies have been conducted on the effects of sediments on corals in the context of understanding the effects of terrestrial run-off and dredging, and the studies have been collated and reviewed many times (i.e., Erfteimeijer et al., 2012; Fabricius, 2005; Foster et al., 2010; Jones et al., 2015b; Rogers, 1990). These studies have contributed to our collective understanding and concern of the environmental effect of sediments on coral reefs. Logically, information from these studies should be suitable for threshold development. However, the results are dependent on the exposure scenarios and experimental conditions used, and it is noticeable that although a great deal of data have been collected (see, Falkenberg and Styan, 2014), there is little information in the public literature about the conditions that can occur in situ during dredging around coral reefs (Jones et al., 2015a). Recently, Harris et al. (2014) highlighted the fundamental significance in ecotoxicology of the use or environmentally relevant or realistic conditions, the need for comprehensive justification of any exposure conditions, and the dangers associated with an incomplete understanding.

We first identify, through a conceptual model, the likely exposure pathway(s) associated with the effects of dredging, including how proximal stressors are interlinked and interact with natural stressors. Data from several recent major dredging programmes in reefal areas of tropical Australia are then examined to describe how the water quality conditions change over time to provide a first order approximation of the environmentally relevant or realistic exposure scenarios. The existing literature of the effects of sediments on corals is then examined with specific attention to the experimental conditions used in order to assess the suitability of past studies for deriving information for impact prediction purposes (i.e. for modelling and monitoring). One of the objectives of the review is also to facilitate a technical understanding of sediments and sediment resuspension, water quality issues associated with dredging programmes, and some of the terminologies used and processes involved. The analysis concludes with some recommendations and guidance for future study.

## 2. Natural and dredging-related turbidity events

Turbidity is the optical property of a suspension that causes light to be scattered and absorbed rather than transmitted through the water column (Davies-Colley and Smith, 2001); it implies muddiness and lack of clarity and transparency (Kirk, 1994). The attenuation of light is related to the suspended-sediment concentration and the water itself, plankton, and other microscopic organisms and coloured organic matter (humics, Kirk, 1994). Suspended-sediments are often the dominant factor affecting light attenuation in tropical and sub-tropical waters.

Natural turbidity events are common in the shallow reef environment and although resuspension and transport of suspended material may be strongly influenced by unidirectional currents, wind-driven waves are the primary mechanism of turbidity generation (Jing and Ridd, 1996; Larcombe et al., 2001; Lawrence et al., 2004; Ogston et al., 2004; Verspecht and Pattiaratchi, 2010). The underlying hydrodynamic principles associated with natural turbidity generation are illustrated in Fig. 1 and further information on hydrodynamic controls of sediment resuspension and sedimentation are given in Larcombe and Woolfe (1999) and in a suite of more recent studies in Hawaii (Ogston and Field, 2010; Ogston et al., 2004; Storlazzi and Jaffe, 2008; Storlazzi et al., 2004).

Briefly, water moves in circular motion with the diameter of orbit equal to the wave height (Van Rijn et al., 1990) and the orbital motion is transferred into deeper water with the radius of the motion decreasing with increasing depth and compressing into ellipses in shallow water (Perry and Taylor, 2009). When tidally-driven near-bottom currents and wave-orbital nearbed speeds are sufficient to exceed a critical bed shear stress seabed erosion occurs, releasing sediment into the overlying column as a suspended load (Ogston et al., 2004; Storlazzi et al., 2004; van Rijn, 2007). Once sediment is suspended a small rise in shear stress can result in a disproportionate increase in suspended-sediment concentration (SSCs) (Orpin et al., 2004). Sediments will remain in suspension until the fluid velocity is insufficient for turbulent eddies to balance gravitational forces and the particles will settle out, depositing on the seabed (Masselink et al., 2014). This could occur in quiescent, calm areas (i.e. embayments), in waning periods after storms (Storlazzi et al., 2009; Verspecht and Pattiaratchi, 2010) or during slack water between tides. Fluctuating shear stresses can also result in successive resuspension and deposition events and if the amount of sediment deposited on the bed exceeds the amount eroded over the same time period then net deposition will occur (McAnally and Mehta, 2001; Ogston et al., 2004).

The phrase 'turbidity-generating event' is used here generically to describe dredging and marine construction related activities which release sediments into the water column. These include release of sediment at the seabed by the action of the drag head or cutter suction head (Fig. 2), release of sediments at the surface by the overflow of fine material from the dredge hopper or hopper barge, spillage from the bucket of a back hoe dredge, and also dredge material disposal. Release from the overflow is one of the primary mechanisms and sediment concentrations in the hopper can measure in the tens of  $\text{g L}^{-1}$  range; however, there is usually an abrupt, rapid initial dilution by a factor of 10–100, with SSCs decreasing logarithmically with time and distance (Duclos et al., 2013; Spearman, 2015; Spearman et al., 2007).

The terms 'near-field' and 'far-field' are used to distinguish sediment plumes based on the behaviour of the resuspended particles and this is closely related to the proximity to dredging (for a detailed discussion see VBKO (2003)). Near-field plumes are also called 'active' plumes and behave in a 'dynamic' manner, with material descending rapidly to the seabed as a density current. Entrainment of ambient water into the plume can slow its descent and entrainment of air bubbles into the plume on discharge and action from the ship's propellers can also lift a portion of the plume to the surface (Fig. 2, see also Jones, 2011b).

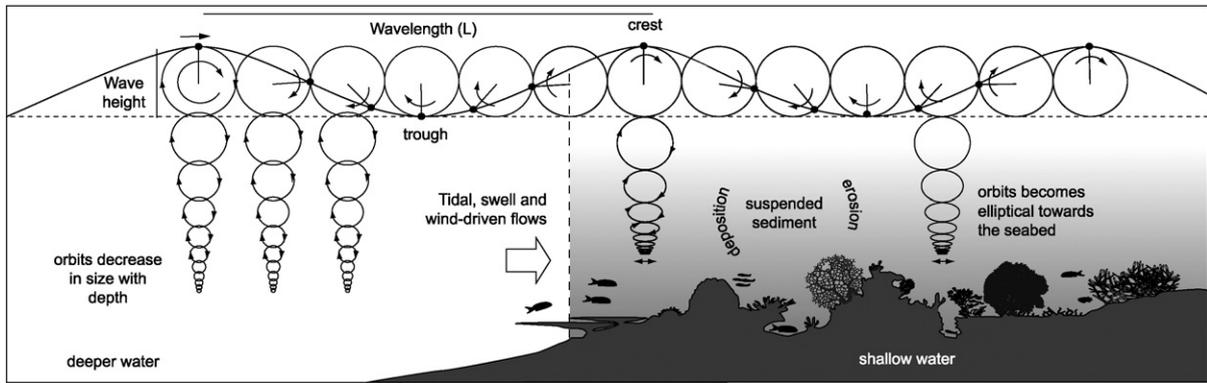


Fig. 1. Stylized representation of current and wave-induced turbidity generation in the shallow, reef environment (see text for explanation).

Settling of coarser rather than finer particles occurs as sediment is advected away from the dredging site and sands will drop quickly within the first 10–20 min (Duclos et al., 2013). The far-field or passive plume is made up of much lower SSCs and the fine fraction of material mixed into the water column can stretch for many kilometres and persist for several hours, as determined by site-specific hydrodynamics and local conditions (Fig. 3B).

3. Conceptual model of the effects of sediments on corals

A conceptual model of the effects of turbidity-generating events on corals based on the US Environmental Protection Agency (USEPA) causal/diagnosis decision information system framework (CADDIS)(Norton et al., 2009; USEPA, 2004) is shown in Fig. 4. In the model, all known cause–effect linkages, biologically plausible cause–effect pathways, and potentially confounding variables and interacting factors are displayed in a single framework. Sources of sediments include carbonate and igneous and siliciclastic intertidal and subtidal sediments released into the water column (Fig. 4). There are many activities associated with dredging and land reclamation activities that can cause turbidity ranging from major sources such as hopper overflow to more trivial sources such as rock-dumping (Koskela et al., 2002). Damage associated with dredging and turbidity-generating activities can be separated into direct and indirect effects (Fig. 4), with direct effects including the removal of hard and soft substrate (within the dredge footprint), and smothering of the seabed at disposal (placement) sites.

A second and much larger group of cause–effect pathways are associated with mobilization of sediments into the water column, i.e. ‘turbidity’ or ‘plume-generation’ and subsequent movement of sediment out of the immediate dredging or dredge material placement site.

These ‘indirect’ effects can be further divided into (1) chemical and (2) physical effects. Chemical effects include the release of FeS-rich sediments which can rapidly deoxygenate water and change pH levels (i.e. a proximal stressor) (Morgan et al., 2012). Desorption of legacy contaminants and release of nutrients and pore-water dilution can also change the chemical environment of the water column (Eggleton and Thomas, 2004; Su et al., 2002). Nutrient release has the potential to change oxygen concentrations, mediated through phytoplankton and microbial blooms (i.e. a step in the causal pathway, Fig. 4). Changes in oxygen and nutrient levels and especially contaminant concentrations have the potential for acute and chronic toxicological, cellular and physiological effects, including genotoxic (mutagenic, teratogenic and carcinogenic) effects, as well as bioaccumulative effects through uptake and ingestion of contaminants (see for example Hedge et al. (2009)). Prior to dredging, sediments are normally examined for contaminant concentrations and, in Australia at least, landfilled if levels exceed screening guidelines (DEWHA, 2009). Many capital dredging projects in the tropics also occur in green-field sites without historical pollution, and sediment contamination is typically less of a concern than industrialized areas (with a few notable exceptions, see Jones, 2011a). Whilst not discounting the potential significance of chemical effects, especially in the near-field environment, the rest of the review is concerned with indirect effects of physical proximate stressors associated with turbidity generating events and include (1) elevated suspended-sediments, (2) changes in light quality and quantity, and (3) sediment covering (Fig. 4).

For all proximate stressors the key interacting factors are periods of naturally elevated turbidity associated with (1) currents, wind-driven waves and elevated sea states associated with trade winds and storms (Jing and Ridd, 1996; Larcombe et al., 2001; Lawrence et al., 2004; Ogston et al., 2004; Orpin and Ridd, 2012; Verspecht and Pattiaratchi,

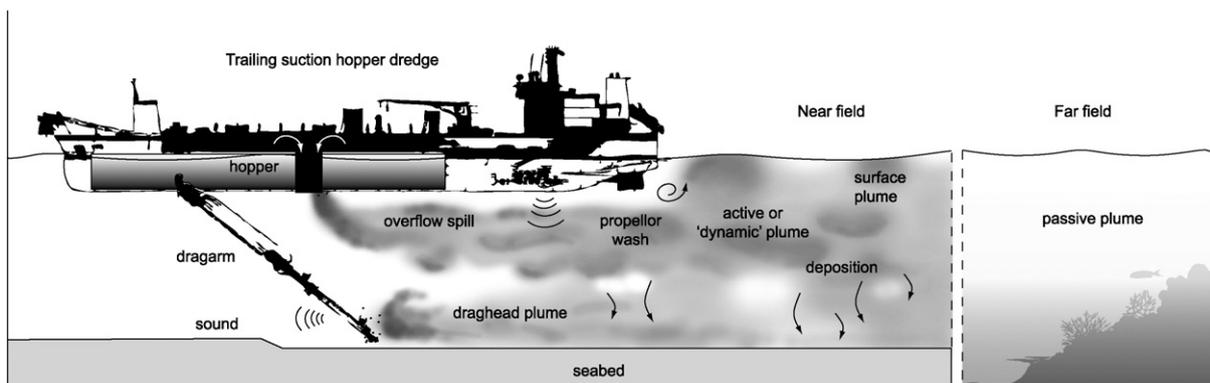
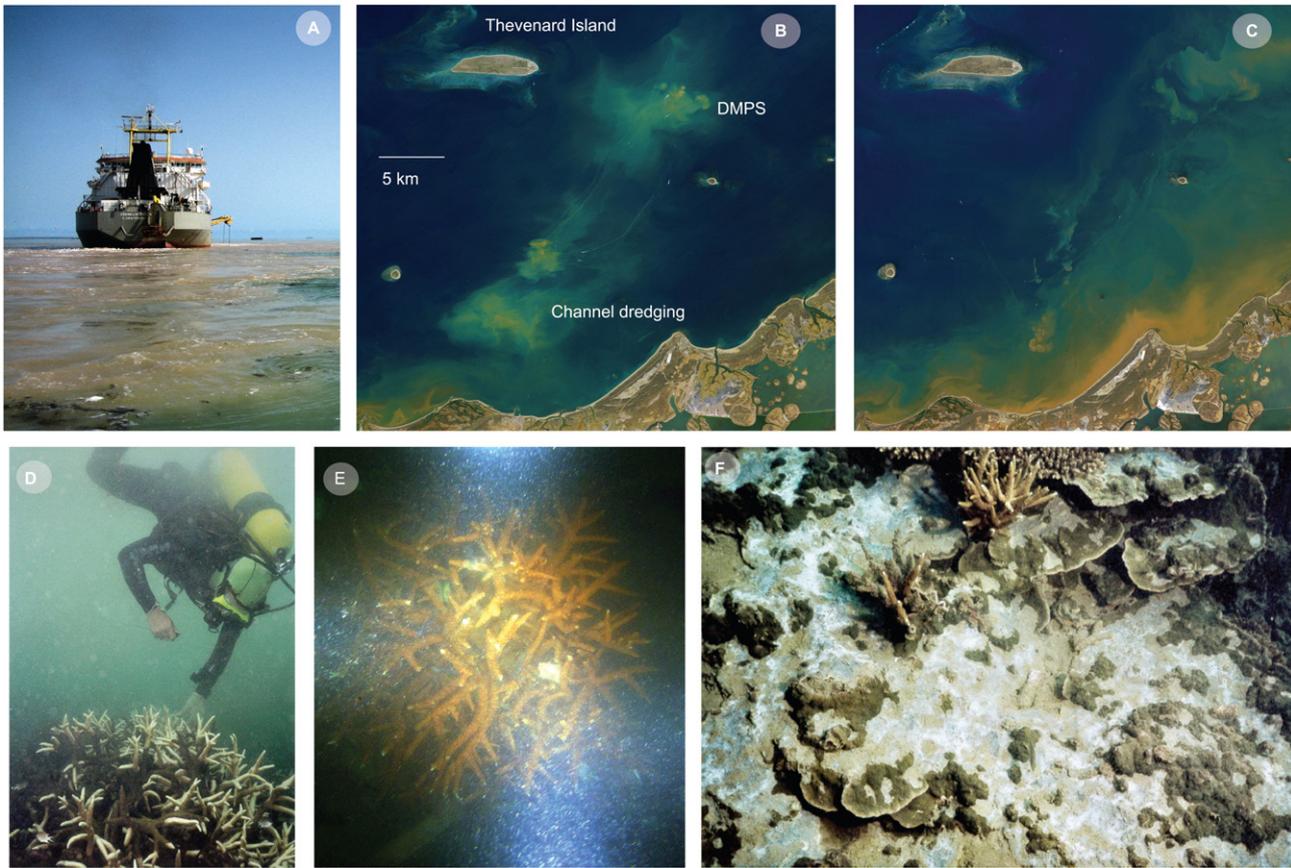


Fig. 2. Stylized representation of plume generation by a trailing suction hopper dredge, the most commonly used of the hydraulic dredges for soft sediments (see Foster et al., 2010). Turbidity is generated at the drag-head and at the surface if sediment-laden water inside the hopper is allowed to discharge (overflow). This can occur from the ship’s sides but usually through a vertical shaft inside the hopper, exiting below water level (Foster et al., 2010; VBKO, 2003).



**Fig. 3.** (A) Working trailing suction hopper dredge in the Pilbara region of Western Australia generating buoyant surface plumes through propeller wash. (B, C) Atmospherically corrected, colour corrected, pan sharpened satellite image from the United States Geological Survey (USGS) Operational Land Imager (OLI) instrument on (B) 10 July 2014, showing sediment plumes caused by dredging and dredge material placement near Onslow in the Pilbara region of Western Australia (where DMPS = offshore dredge material placement site) and (C) 25 September 2013, showing natural turbidity caused by a high wind event. (D) Buoyant turbidity plume overlying a branching *Acropora* spp. colony. (E) A branching *Acropora* spp. colony photographed during the daytime, and temporarily in darkness caused by the elevated turbidity levels. (F) Smothering of a plating *Montipora* spp. by sediment. Images D and F were taken during a dredging project at Magnetic Island (Central Great Barrier Reef) and image E during the Barrow Island project. Satellite images courtesy of Mark Broomhall and Peter Fearn (Department of Imaging and Applied Physics Curtin University of Technology).

2010) and (2) in the nearshore, coastal environment, the effects of river-borne discharges from monsoonal events and tropical depressions, and associated sediment-laden, hypopycnal flood waters (Orpin and Ridd, 2012; Storlazzi et al., 2009). Natural cycles of light availability are also an interacting factor and summarized in Anthony et al. (2004), these include (1) the seasonal pattern of daily surface irradiance (insolation) governed by the solar declination cycle (Kirk, 1994), (2) tidal cycles which affect the depth of the water, (3) meteorological phenomena including patterns of cloud formation (Wright, 1997) and (4) large-scale pressure systems such as the Madden-Julian oscillation (MJO) (Madden and Julian, 1994).

#### 4. Cause-effect pathways

##### 4.1. Suspended-sediment

Corals are both autotrophs and heterotrophs and many studies have shown raptorial capabilities and the capture of up to meso/macro sized zooplankton by nematocyst discharges and tentacle grabbing (Ferrier-Pages et al., 2003; Marshall and Orr, 1931; Sebens et al., 1998; Sebens et al., 1996; Vaughan, 1916). Zooplankton feeding contributes significantly to fixed carbon incorporated into coral skeletons (recently reviewed by Houlbrèque and Ferrier-Pages (2009)).

From the early suggestions of Goreau et al. (1971) and Muscatine and Porter (1977), corals have been shown to gain some energy heterotrophically by tentacular suspension feeding (Houlbrèque and Ferrier-

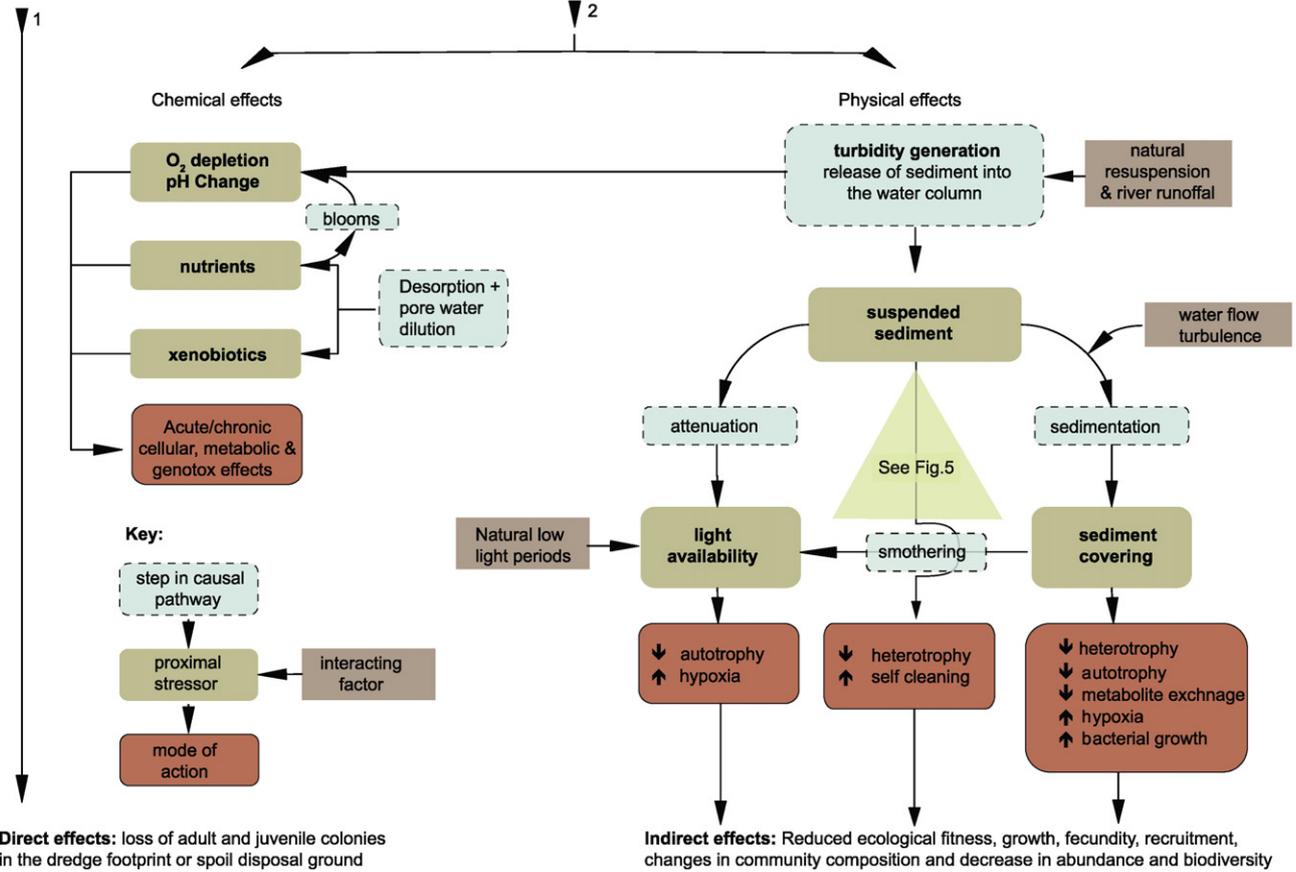
Pagès, 2009). General suspension feeding mechanisms include direct interception and electrostatic attraction (LaBarbera, 1984; Rubenstein and Koehl, 1977) and once captured there is a second stage involving particle retention and a third phase involving movement of captured particles to the mouth (Shimeta and Koehl, 1997). Tentacular suspension feeding for particulate matter occurs through entrapment in mucus on the coral surface. Another capture process is via mucus threads or filaments attached to the oral disk that are swept by water turbulence entangling fine particulate material as well as larger zooplankton (Lewis and Price, 1975, 1976; Lewis, 1976). Ingestion is completed by movement by cilia of particles trapped in the mucus to the mouth (Lewis and Price, 1975; Vaughan, 1916).

Recent studies have shown that the coral cilia can generate vertical flows extending up to 2 mm from the surface which can substantially enhance the transport of solutes to and from the coral surface (Shapiro et al., 2014) suggesting corals may be complementing passive entrapment by active mechanisms. Marshall and Orr (1931) observed that ciliary movement of deposited sediment is a short term response which cannot be maintained for long periods, implying a metabolic cost. However, the energy invested in powering the ciliary movement is reported to be a negligible fraction of the coral's metabolic budget (Shapiro, 2014), but how this changes in response to increased drag associated with high SSCs has yet to be determined.

Ingestion of sediments has been observed in many studies (Lewis and Price, 1975; Lewis, 1976; Logan, 1988; Marshall and Orr, 1931; Stafford-Smith and Ormond, 1992; Stafford-Smith, 1993) and appears

**Source:** Marine and/or terrestrially derived siliclastic and/or mixed biogenic subtidal and intertidal marine sediments

**Human activity:** Capital and maintenance dredging of: shipping channels, gas trunklines, berth pockets, turning basins, material offloading facilities dredge material (spoil) disposal • sand borrowing • rock dumping • land reclamation • bed levelling • propellor wash and other marine infrastructure operations (marina dewatering)



**Fig. 4.** A conceptual model of the effects of turbidity-generating events on corals based on the US Environmental Protection Agency (USEPA) causal/diagnosis decision information system framework (CADDIS), identifying the sources of sediment, dredging-related turbidity-generating activities, all known cause–effect pathways, biologically plausible cause–effect pathways, modes of action, interacting factors, and likely physiological and ecological responses (see text for explanation).

to be part of a normal feeding mechanism. Suspended particulate matter constitutes a potentially diverse food source containing bacteria, microalgae, protozoa, detrital organic matter (Marshall, 1965), interstitial invertebrates, detached, undissolved mucus (Wild et al., 2004), microbial exudates, and excretory products from other animals (e.g. from fish, Meyer and Schultz, 1985; Lopez & Levinton 1987; Houlbrèque and Ferrier-Pagès, 2009). Many studies have now shown that after sediment ingestion corals are capable of assimilating and obtaining nutritional benefits from the associated organic matter (Anthony, 1999, 2000; Anthony and Fabricius, 2000; Mills and Sebens, 1997; Mills et al., 2004; Rosenfeld et al., 1999).

Mills and Sebens (1997) suggested that high loads of clean sediments may cause most of the polyps to stop feeding and reject sediments, reducing ingestion rates. Similarly, Anthony (2000) indicated that at concentrations >30 mg L<sup>-1</sup> *Acropora millepora* and *Pocillopora damicornis* from clear water offshore environments showed a tendency to retract their polyps reducing potential for energy gains from feeding. It follows that depending on organic content, water flow and morphology, low SSCs may be beneficial for some corals in some circumstances and detrimental at higher concentrations. This concept of a low dose stimulation and high dose inhibition has now been reported in numerous studies (Anthony, 1999, 2000; Logan et al., 1994; Mills and Sebens, 1997; Mills et al., 2004; Mills and Sebens, 2004; Rosenfeld et al., 1999; Stafford-Smith and Ormond, 1992; Tomascik and Sander, 1985). Whilst low suspended-sediment concentrations can have beneficial effects,

overall high SSCs is one of the key pressure parameters reducing feeding activity and requiring energy to continually process and transport intercepted sediments (Fig. 4).

#### 4.2. Light availability

Most reef-building corals form mutualistic symbioses with dinoflagellates of the genus *Symbiodinium* (Freudenthal, 1962), a diverse range of microalgae divided into nine known clades with significant functional and genetic intercladal diversity (Stat et al., 2012). The algae provide 'photosynthates' or photosynthetically fixed carbon to the host, providing additional energy for respiration and growth (Dubinsky et al., 1984; Falkowski et al., 1984; Goreau, 1959; Lesser, 2004; Muscatine, 1990). The *Symbiodinium* spp. reside endosymbiotically in the coral endodermal (gastrodermal) tissues within a membrane complex, the symbiosome (Roth et al., 1988; Wakefield and Kempf, 2001), at densities of typically one to two, but sometimes up to six per host cell (Muscatine et al., 1998). Forming the symbiosis enhances deposition of the calcium carbonate skeleton in light-enhanced (DCMU-sensitive) calcification (Gattuso et al., 1999). The highly reflective skeleton also enhances the light field experienced by a polyp through light scattering and diffuse reflection, increasing the probability of absorption and increasing the exposure to photosynthetically active radiation (PAR) by 3–20 fold (Enriquez et al., 2005; Kuhl et al., 1995; Marcelino et al., 2013; Reef et al., 2009).

Corals and their algal symbionts are superbly adapted to living in both low light and high light environments, exhibiting behavioural morphological and physiological plasticity to maximise light utilization and minimize damage (Roth, 2014). For the host (animal) these include changes in polyp retraction (Levy et al., 2003) and micro- and macro-scale growth morphology (Barnes, 1973). For the algal symbionts this includes changes in optical cross sectional areas and light harvesting capabilities through changes in photosynthetic and accessory pigment concentrations (Anthony and Hoegh-Guldberg, 2003; Falkowski and Dubinsky, 1981; Falkowski et al., 1984; Mass et al., 2010).

The quantity and quality (spectral composition) of the submarine light field is fundamentally important for the physiology and ecology of the coral–algal symbiosis and light attenuation, mediated by absorption and scattering of light by suspended particles (i.e. a step in the causal pathway), is one of the key proximal stressors in the short and especially long-term. In very low or zero light conditions, corals can enter a state of hypoxia and then anoxia; the mode of action is through reduced autotrophy and hypoxia (considered further below).

#### 4.3. Sediment covering

The immediate response of corals to deposition of sediments on their surfaces is an attempt to self-clean by moving sediments to edges where they are dropped off the colony (Marshall and Orr, 1931). The principal sediment rejection mechanisms identified for a range of coral species, representing different families and a range of growth forms and corallite morphologies are: ciliary action, hydrostatic inflation, tentacle movement, contractions, and mucus entrapment (Bak and Elgershuizen, 1976; Hubbard and Pocock, 1972; Logan, 1988; Marshall and Orr, 1931; Schuhmacher, 1977; Stafford-Smith and Ormond, 1992; Stafford-Smith, 1993; Vaughan, 1916; Yonge and Nicholls, 1931). These are usually referred to as active processes (requiring energy) and most corals employ ciliary action and hydrostatic inflation (sometimes in conjunction with pulsed contractions and polyp expansion), mucus entanglement, and tentacle movement to remove sediments.

A number of other mechanisms have been proposed including movement of sediment by mesenterial filaments, capture of sediment by nematocysts, and sediment ingestion. Commensal crabs become more active during high sediment deposition rates (Stafford-Smith and Ormond, 1992) and can create local turbulence, which dislodges particles. Turbulence from feeding fish can have similar effects (Loya, 1976). It is unclear whether some of these proposed mechanisms are associated with feeding responses and whether they represent physiologically significant sediment-rejection mechanisms. Observations of tentacle movement and processing of individual sediment particles may be a consequence of the large, sand-sized particles used in some of these feeding experiments (Stafford-Smith and Ormond, 1992). Ingested sediments are subsequently regurgitated several hours later as mucous-bound pseudo faeces (Logan, 1988), so sediment ingestion will not clear sediment from the surface.

Corals also have gross and fine scale skeletal morphologies which can assist gravitational forces to remove sediments from their surfaces. These include surface inclination, branch spacing and diameter in arborescent species and aspect ratio and the degree of sphericity in massive species. Finer scale morphology relates to calical size and shape (Hubbard and Pocock, 1972; Lasker, 1980; Logan, 1988; Marshall and Orr, 1931; Vaughan, 1916). Stafford-Smith (1993) also proposed a new parameter, surface smoothness, which is an index of surface microarchitecture and the ability of corals to expand tissues (by the active process of hydrostatic inflation) above skeletal projections, resulting in a shape which better sheds sediments passively. Calix inclination is a key parameter Logan (1988).

Once the sediment clearance rates have been exceeded, sediments will inevitably build-up on a coral's surface and it becomes progressively buried in a sediment deposit (i.e. smothering, Fig. 3F). The

ultimate fate of the underlying tissues is partial mortality (lesion formation), unless the layer is removed by a storm. One of the significant issues associated with smothering is tissue hypoxia, brought about by either light attenuation caused by a sediment covering or a reduction in gas (solute) transfer across diffusive boundary layers (DBLs). Corals are oxygen conformers, routinely experiencing pronounced diel changes in tissue oxygen concentrations ranging from super-saturation (hyperoxia) during the daytime associated with algal photosynthesis, to night-time oxygen shortage (hypoxia) or even anoxia by host and algal respiration (Jones and Hoegh-Guldberg, 2001; Kuhl et al., 1995; Shashar et al., 1996). In darkness, or low flow conditions, oxygen concentrations can fall to levels where aerobic respiration and ATP generation is limited, a situation analogous to flooding of terrestrial plants (Bailey-Serres and Voeselek, 2008; Fukao and Bailey-Serres, 2004). Sea-anemones can survive hypoxic periods by engaging in fermentation processes involving glycolysis (Ellington, 1977; Ellington, 1980, 1982) which generates some ATP although at much lower yields (6× less) than aerobic respiration (Shick, 1991). Fermentation processes have been regularly implicated in corals as a means of tolerating short-term hypoxia (Weber et al., 2012; Wooldridge, 2013).

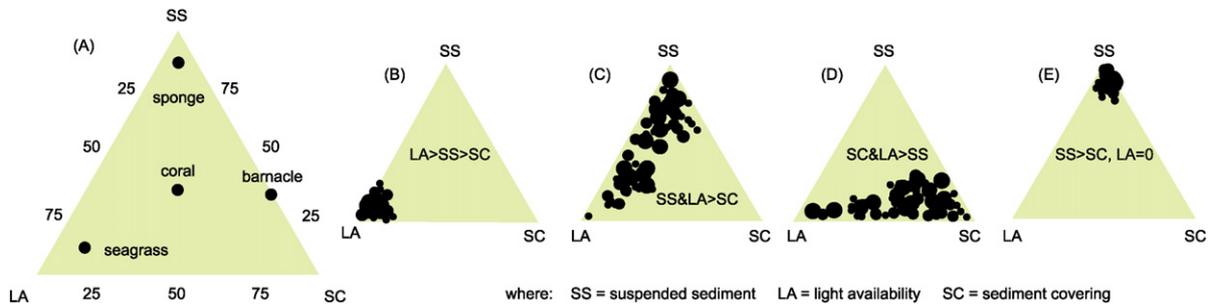
The mechanism underlying sediment-smothering induced mortality and local necrosis has been described in detail by Weber et al. (2012). A rapid (<24 h) microbially-mediated anoxia and change in pH was recorded in *Montipora peltiformis* smothered with a couple of millimetres of organically-rich sediments, leading to localised necrosis. Sediment smothering is therefore a key pressure parameter associated with turbidity generation, resulting in boundary-related effects and decreased solutes (such as oxygen) and metabolite exchange, mass transport limitations, and decreased filtering/feeding. Sediment smothering is related to sedimentation processes in the water column, which is dependent on wave movement turbulence and local hydrodynamics (i.e. an interacting factor).

#### 4.4. Combined effects of proximal stressors

The key pressure parameters identified above and associated with the indirect effects of dredging activities on reef communities (represented by the inner triangle in Fig. 4) are: (1) high SSCs affecting feeding (heterotrophy reduction), (2) a reduction in light and associated effects on the photosynthesis of the symbiotic microalgae of corals (phototrophy reduction), and (3) sediment smothering which causes a reduction in gas (solute) transfer across diffusive boundary layers. These three cause–effect pathways are highly interconnected, with suspended-sediments causing biological effects directly, but also acting as a causal step to changes in light quality and quantity (through attenuation and scattering in the water column). Similarly, high SSCs are a prerequisite for sediment deposition, mediated by the process of sedimentation (a casual step) in the water column. Once the coral's surface is veneered or smothered by sediment, the effect will be similar to light reduction in the water column and feeding processes will also be affected, and for this reason smothering by sediments has most biological effects associated with it (Fig. 4).

The relative influence of these three key pressure parameters can be symbolized in the ternary diagrams of Fig. 5 with epibenthic filter feeders such as sponges most likely to be more affected by high SSCs (Bell et al., 2015) and sitting at the apex of the triangle (Fig. 5A). Seagrasses, which have comparatively high light requirements (Dennison et al., 1993), are likely to be located more towards the lower left hand side. Non-photosynthetic benthic filter feeders, such as tropical barnacles (Fabricius and Wolanski, 2000), are likely to be located on the right hand-side. Species with mixed modes of nutrition, such as symbiotic hard and soft corals, are likely to be equally influenced and are represented at the triangle's orthocentre (Fig. 5A).

During dredging, these representations are likely to move on an hourly, daily, and seasonal basis, depending on the dredging activities, diel and tidal cycles, and sea-state. Four different exposure



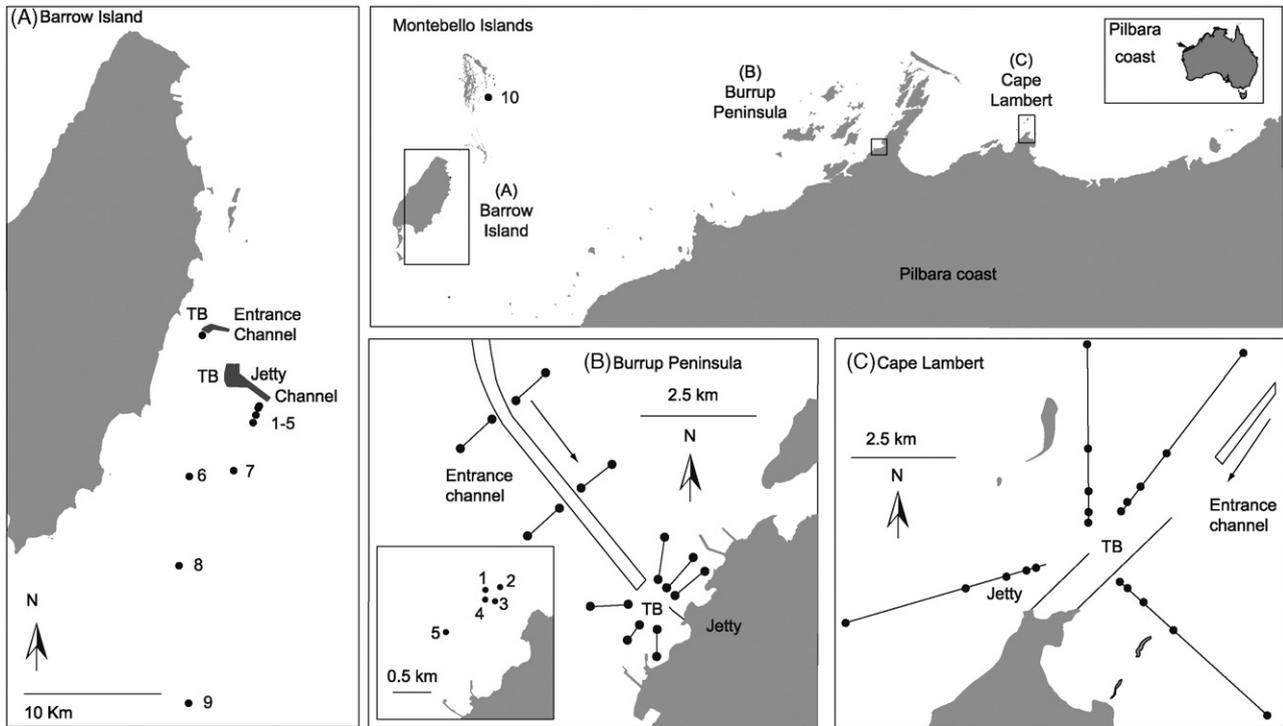
**Fig. 5.** Ternary diagrams representing the relative influence of 3 key proximal stressors associated with turbidity generation activities i.e. suspended sediment (SS), light availability (LA) and sediment covering (SC) on shallow tropical benthic organisms such as corals, sponges, seagrass and filter feeders such as barnacles. During dredging the relative influence of the proximal stressors could move on an hourly, daily, and seasonal basis, depending on the dredging activities, diel and tidal cycles, and sea-state. Four different exposure scenarios for symbiotic corals exposed to dredging plumes are symbolized: (B) A scenario whereby a buoyant plume drifts over the reef with little contact with the corals and where light availability only is likely to be the key proximal stressor (see photograph Fig. 3D), (C) A turbidity event during relatively turbulent water conditions where (wave + current) shear stresses are sufficient to inhibit the deposition of most sediment and so suspended-sediments and reduced light availability are the predominant influences (see photograph Fig. 3E), (D) A scenario where elevated SSCs has occurred during very calm conditions and where sediment has subsequently fallen out of suspension and smothered corals (see photograph Fig. 3F), (E) represents a night time scenario of high SSCs in turbulent conditions where suspended-sediment is the sole proximal stressor.

scenarios for symbiotic corals exposed to dredging plumes are represented in Fig. 5B–E. In a scenario of a buoyant plume drifting over the reef with little contact with the corals, light availability is likely to be the key stressor (Fig. 5B and photograph Fig. 3D). Under energetic water conditions, where most sediment is in suspension, suspended-sediments and reduced light availability are the predominant influences (Fig. 5C and photograph Fig. 3E). Fig. 5D represents a scenario where elevated SSCs occurred during calm conditions and where sediment has fallen out of suspension and smothering has occurred (Fig. 3F). Fig. 5E represents a scenario of high SSCs in turbulent conditions at night

time, where suspended-sediments are the predominant influence while light attenuation and reduced light availability has no influence.

**5. Water quality characteristics of dredging plumes**

There have recently been several major dredging programmes in Australia where the state and federal regulatory conditions have required detailed water quality monitoring programmes over extended periods using arrays of in situ instrumentation (Fig. 6, Falkenberg and Styan, 2014). These data have been made available for study and are



**Fig. 6.** Location map showing the Pilbara coast in Western Australia and the sites of 3 recent large scale capital dredging projects, including: (A) the Barrow Island project (~7.6 Mm<sup>3</sup> of sediment dredged over 530 days from 19 May 2010 to 31 October 2011, Ministerial statement 800), (B) the Burrup Peninsula project in the Dampier Archipelago (~12.5 Mm<sup>3</sup> of sediment dredged over 911 days from 22 Nov 2007 to 21 May 2010, Ministerial statement 757), and (C) the Cape Lambert Dredging Project (~14 Mm<sup>3</sup> of sediment dredged over 633 days from 22 Dec 2007 to 15 Sept 2012, Ministerial statement 840). These projects were associated with dredging for entrance channels, approach areas, turning basins (TB), material offloading facilities, berth pockets, and liquefied natural gas (LNG) jetties and tug harbours and involved deepening the seabed to –12.5 to –20 m to allow access for bulk iron ore and LNG transport vessels. The dredging campaigns generally occurred on a 7 days × 24 h basis, stopping only for maintenance and bunkering requirements, with most projects using a combination of trailing suction hopper dredges, cutter suction dredges and back hoe dredges, and bed levellers, and with dredge material placement at offshore disposal sites. The ministerial approval statements (MS) for these projects are searchable on the WA EPA website ([www.epa.wa.gov.au](http://www.epa.wa.gov.au)).

examined below in terms of defining environmentally relevant or realistic conditions during dredging programmes.

### 5.1. Particle size distributions (PSDs) associated with dredging

Sediment PSDs in the water column were measured during the Burrup Peninsula dredging programme in Western Australia (Fig. 6). Samples were collected inside a plume 125–200 m away from an operating trailing suction hopper dredge and at a reference site outside of the plume and ~800 m southwest of the dredge (Fig. 6B insert). Sampling inside the plume occurred over a 65 min period and at the reference site outside of the plume a further 45 min later. Triplicate water samples were collected at the top, middle and bottom of the water column using a 2.5 L Niskin bottle. Samples were filtered onto Whatman 47 mm GF/F filters, and 100 mL of distilled water used to rinse the container, filter funnel and filter pads of salts. Filters were then dried overnight in a 65 °C oven and weighed. Water column PSDs were assessed using a Laser In-Situ Scattering and Transmissometry (LISST) 100X Type C (Sequoia, WA, US) calculated by examining the angular distribution of forward scattered light (referenced to pure water) over the range from 2–500 µm using the proprietary inversion process. Data shallower than 0.3 m were discarded to remove the possibility of artefacts from small bubbles and dissolving salts from the detector window upon immersion.

Gravimetrically determined depth-averaged SSC ranged from 12–110 mg L<sup>-1</sup> within the plume, with the highest SSCs recorded at the surface and the seabed (Fig. 7A) and with depth averaged PSDs ranging within the silt-sized (4–62.5 µm) fraction. Surface plumes had comparatively finer PSDs <62.5 µm, but peak values of ~60 µm were observed in bottom samples, collected 1 m from the seabed (Fig. 7A). PSDs at the reference site were fine- and medium-sized silts (Fig. 7A). Similar measurements at the offshore spoil ground showed lower SSCs and much smaller particle sizes, with peaks between 10–20 µm and a pattern of slightly coarser sediments at the seabed (Fig. 7C). PSDs of the seabed collected before the start of the dredging programme (see below) indicated a mixture of sand (22%), silt (38%), and clay (40%) was being dredged at the time of sampling.

Grain size analysis of surficial sediments within the area influenced by dredging plumes was examined a few months before and after both the Burrup Peninsula and Cape Lambert projects (Fig. 6). For the Burrup Peninsula project, surficial (top 10 cm) sediment samples were collected by SCUBA divers at ~50 sites along 8 transects with 4 nearshore transects (0.5–2 km from shore) and 4 offshore transects (2–4 km from shore) ranging from 0.1–1 km from the edge of the dredged channel or turning basin (Fig. 6B). Sediments were analysed by a commercial laboratory using Australian Standard (AS) 1289.3.6.2 and 1289.3.6.3 and data expressed as relative percentage of particle sizes in each of the four classes: gravel (>2000 µm), sand (2000–62.5 µm), silt (6.25–2 µm), and clay (<2 µm). For the Cape Lambert project, 5 surficial (top 10 cm) sediment cores were collected by SCUBA divers at ~20 sites along 4 transects (Fig. 6C), with 5 sites along each transect located at distances of approximately 0.25–5 km from the edge of the turning basin (Fig. 6). Sediments were analysed by a commercial laboratory using wet sieving techniques for samples >500 µm and laser diffraction (Malvern Instruments Mastersizer MS2000) for fractions between 2 and 500 µm (ISO 13320-1) and data expressed as relative percentage of particle sizes in each of the four classes: gravel (>2000 µm), sand (2000–62.5 µm), silt (62.5–4 µm), and clay (<4 µm).

The pre-dredging surveys in the Burrup Peninsula dredging project indicated difference in the sediment composition according to proximity to the coast, with the nearshore samples (located within ~2 km of the land) composed of approximately equal fractions of sand, silt, and clay whereas the more offshore sediments (>3 km) were overwhelmingly dominated by sand (~70%) (Fig. 8A). Overall, there are clear differences in the sediment composition after the dredging programme, with

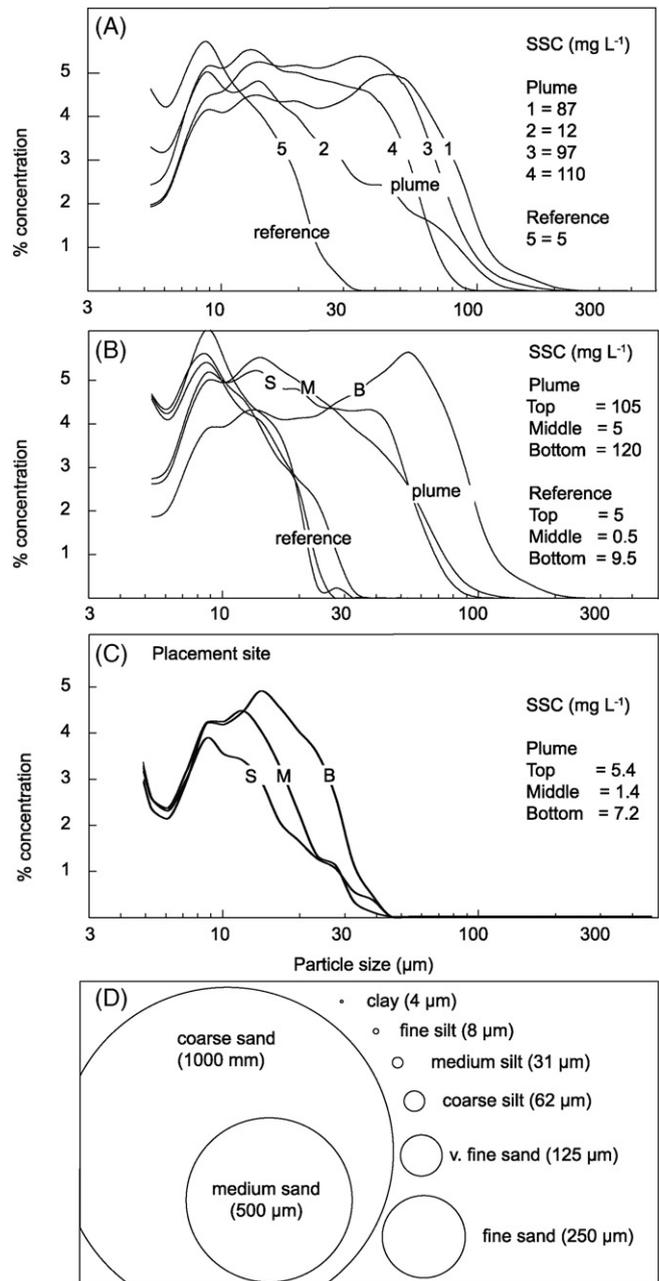
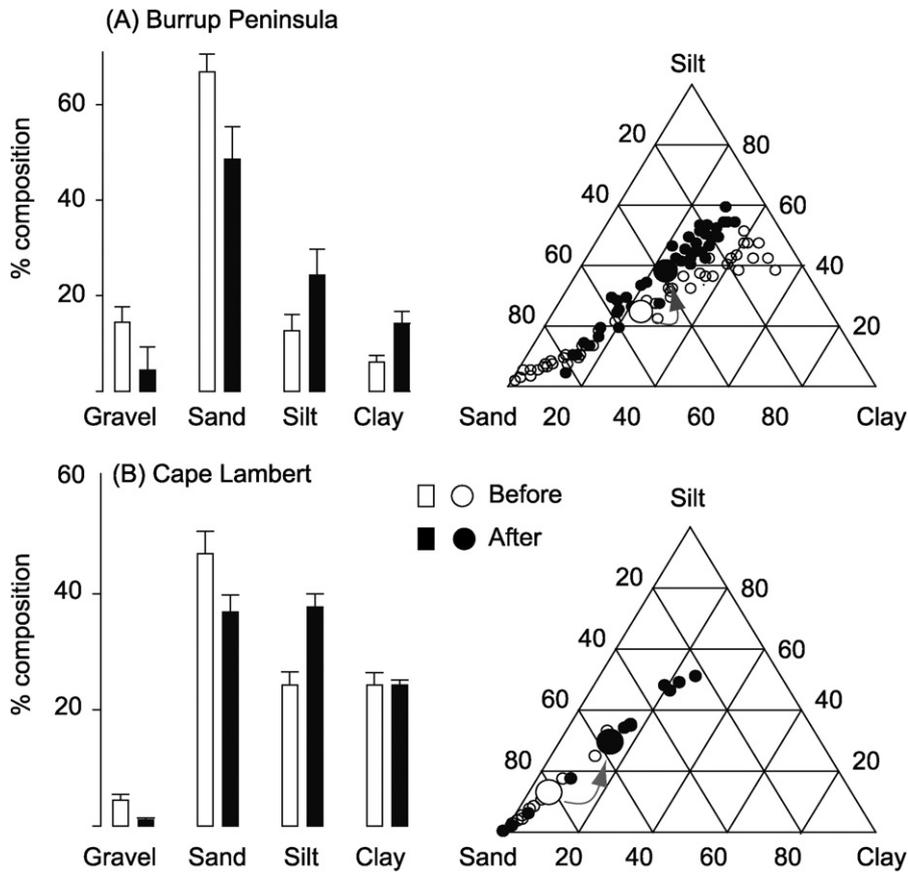


Fig. 7. Sediment particle size distributions in the water column, collected 150 m away from a working trailing suction hopper dredge in a capital dredging programme, showing the fine nature of the released sediments (Burrup Peninsula Project, see Figs. 3A and 6B). (A) Depth averaged particle size distributions (µm) and suspended sediment concentrations (mg L<sup>-1</sup>) inside and outside of a dredging plume. Numbers refer to the sites shown in Fig. 6 B. (B) Average particle size distributions at the top (>0.3 m deep), middle (4–5 m) and bottom (8–9 m or 1 m from seabed) of the water column inside and outside of the plume or (C) at the dredge material placement site. (D) Stylized display of coarse sand through silts to clay based on the Udden–Wentworth US standard classification scale of sediments.

increases in the silt content and decreases in the sand/gravel fraction (indicated as a movement of the larger circular symbols in the ternary diagram of Fig. 8). The second post-dredging survey (conducted 6 months after the first) showed a near identical pattern of sediment composition (data not shown). In the Cape Lambert dredging programme, all sediments were sandy (70%) before dredging, with silts and clays making up only 10% and 5% respectively, of the size fractions. After the 2-year dredging programme, there was a reduction in the gravel and sand fraction of the sediment and a near doubling of the silt and clay fraction to 20% and 10%, respectively (Fig. 8B).

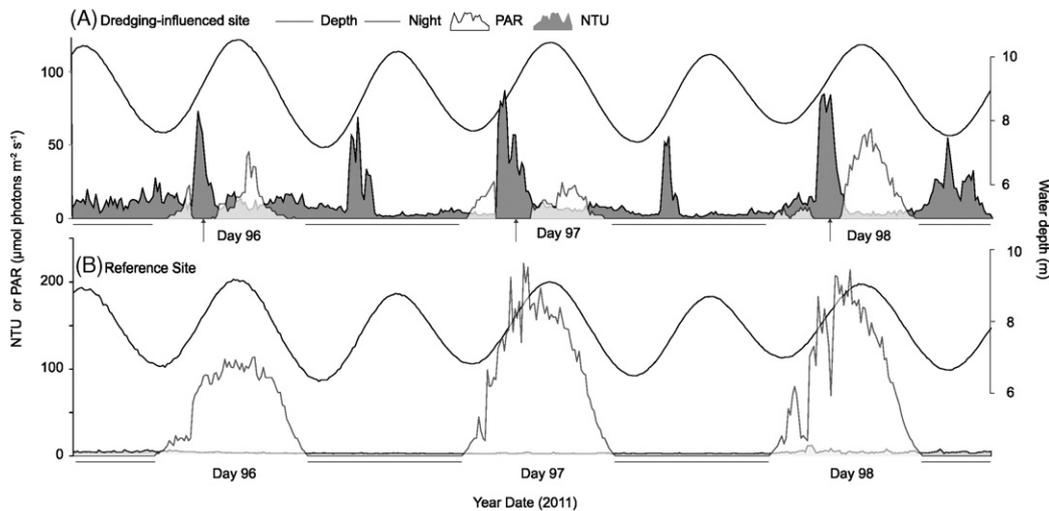


**Fig. 8.** Particle size distributions in surficial sediment before and after dredging, showing a shift to finer grain sizes after dredging. (A) Cape Lambert and (B) Burrup Peninsula dredging projects (see Fig. 6B, C). The accompanying ternary diagrams show clay–silt–sand–gravel size distributions based on the Udden–Wentworth standard classification scale with the larger circles representing the average PSD before and after dredging.

5.2. Suspended-sediment concentrations (SSCs)

At the Barrow Island project (Fig. 6A), turbidity and light levels were recorded on a single sensor platform attached to a steel frame mounted ~40 cm from the seabed. Turbidity was measured using a

single sideways mounted optical backscatter device (nephelometer) and Photosynthetically Active Radiation (PAR) was recorded using a 2π quantum sensor (see Jones et al., 2015a, 2015b for water quality measurements and site descriptions). Data were recorded every 10 min before and during the ~1.5 year dredging programme at



**Fig. 9.** Turbidity, light and water depth during the Barrow Island project close to the dredging and at a distantly located reference site, showing loss of all benthic light during short-term, dredging related turbidity events. Shown are the NTU and instantaneous PAR ( $\mu\text{mol photons m}^{-2} \text{s}^{-1}$ , primary y-axis) and water depth (metres, secondary y-axis) at Barrow Island at (A) site 1 in Fig. 6A and at (B) a reference site near the Montebello Islands (site 10 in Fig. 6) over a 4 day period in April 2011. Black bars represent night time periods and arrows represent daytime darkness periods when elevated SSC levels have reduced PAR levels to  $0 \mu\text{mol photons m}^{-2} \text{s}^{-1}$ . Turbidity at the reference site is barely detectable, averaging  $<1.5 \text{ NTU}$  over the 3 day study period.

multiple sites, but for the purpose of this analysis, data were averaged from 3 of the sites closest to the dredging (see Fig. 6A) located 250 m from where the dredging was occurring.

A characteristic feature of the water quality data is pronounced temporal variability, shown in Fig. 9A for a 3-d period in April 2011 where NTUs underwent short term (several hours) 20-fold excursions, from  $<5 \geq 80$  NTUs. These events were associated with tidal cycles and turbid plumes passing over the sensor platforms on an incoming tide. When the turbidity levels peaked, the benthic light levels decreased to  $0 \mu\text{mol photons m}^{-2} \text{s}^{-1}$  during the daytime (see arrows in Fig. 9A). At a reference location with similar water depth (8–10 m) but located 35 km from the dredging and uninfluenced by sediment plumes, turbidity levels were very low, averaging  $<1.5$  NTU over the 3 day period and mid-day light levels peaked at  $\sim 100\text{--}200 \mu\text{mol photons m}^{-2} \text{s}^{-1}$ . This indicates that the complete loss of daytime light near the dredging was caused by turbidity rather than high cloud cover.

Over the 1.5-year dredging programme turbidity levels regularly exceeded 100 NTUs compared to baseline levels of  $<20$  NTUs, but these episodic periods of poor seawater quality were interspersed with periods of good seawater quality (Fig. 10). Given this characteristic, a running means analysis of the raw 10 min turbidity and light measurement was conducted over multiple incrementing time periods from 1 h (for turbidity) or 1 day (for PAR) to 30 d. Each running time period calculated the average of the previous  $N_T$  data points, where  $N_T$  is the number of samples in the  $T$  hour mean. For example, for the 2-h running mean ( $T = 2$ ),  $N_T = 12$  as there are six 10-min samples per hour. The  $T$  hour running mean at a point in time  $t$

$$\bar{x}_T(t) = \frac{1}{N_T} \sum_{i=1}^{N_T} x_i(t) \quad (1)$$

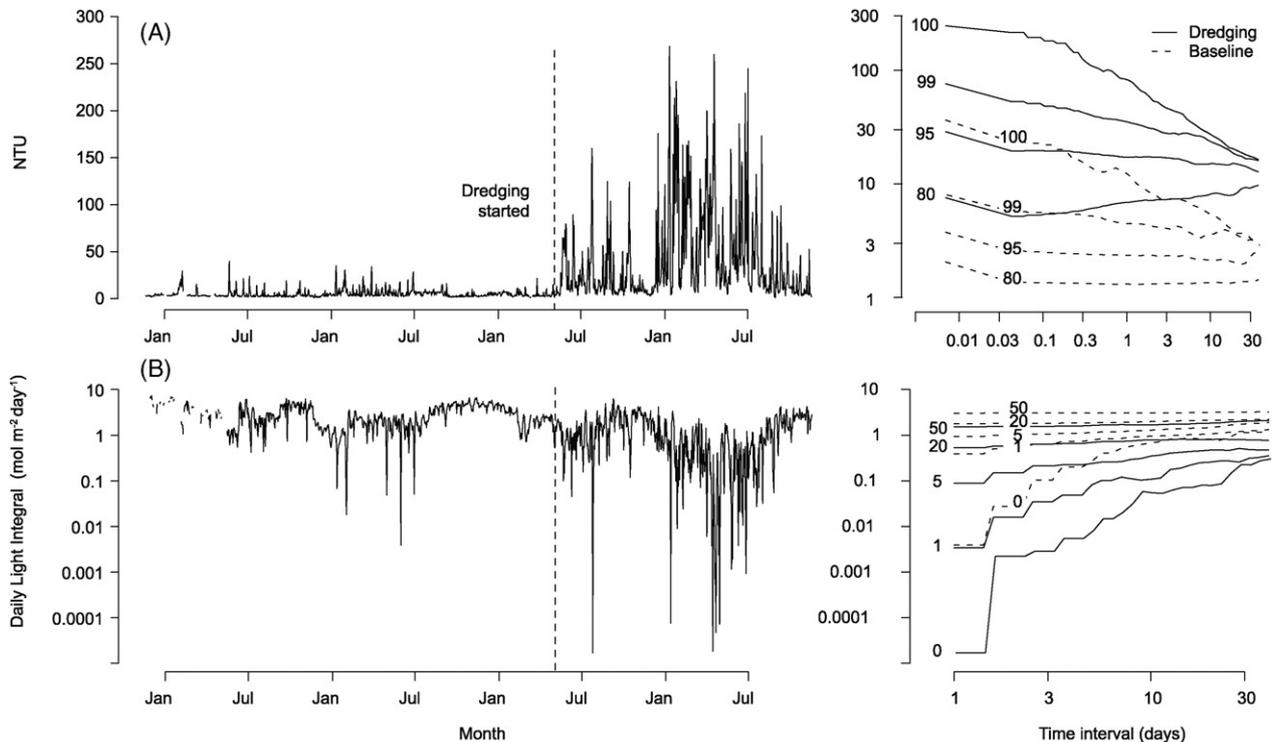
Where  $\bar{x}_T(t)$  is the mean calculated over the previous  $T$  hours of the data from time  $t-T$  to time  $t$  hours, and  $x_i(t)$  are the  $N_T$  data points up to and including time  $t$ . To avoid biased averages, no  $\bar{x}_T$  value was recorded

if more than 20% of the data points for any particular running mean time period calculation were missing. Percentile values of the running mean values  $\bar{x}_T(t)$  for each running mean period were then calculated for the pre-dredging and dredging periods.

In R (R Core Team, 2014), running means were calculated by converting the data series for each site into an S3 time series object using the zoo function from the zoo library (Zeileis and Grothendieck, 2005) then applying the runmean function from the caTools library (Tuszynski, 2013). Once running means for each time span were calculated, these were summarized using an average along with various percentile ( $P$ ) values ( $P_{50}$ ,  $P_{80}$ ,  $P_{99}$  and maximum for turbidity and  $P_{50}$ ,  $P_{20}$ ,  $P_5$ ,  $P_1$  and minimum for PAR). These were plotted as a function of the running mean time span and compared for the pre-dredging and dredging periods. Analysis of the baseline data (i.e. pre-dredging) captures short term, transient increases in turbidity associated with natural turbidity events, whilst analysis when dredging was underway shows the effect of turbidity generating effects on top of the natural patterns.

Prior to dredging the  $P_{95}$  percentile of the 1 h, 1 day, and 1 week turbidity values were all 2–3 NTUs, whilst during dredging these values were over an order of magnitude higher at 22, 18, and 16 NTUs, respectively (Fig. 10A). The 5th percentile of the daily light integral (DLI, mol photons  $\text{m}^{-2}$ ) for 1 day and 1 week running mean periods was  $\sim 0.9$  and  $1.3$  mol photons  $\text{m}^{-2}$  in the baseline period but 0.1 and 0.3 mol photons  $\text{m}^{-2}$  when dredging was underway (Fig. 10B). During the dredging programme one of the worst sequences of light reduction was a 16-d period in the first few weeks of May 2011, where the average DLI did not exceed  $0.1$  mol photons  $\text{m}^{-2}$  and the maximum instantaneous PAR was  $30 \mu\text{mol photons m}^{-2} \text{s}^{-1}$ .

Daily light integrals averaged  $3.3$  mol photons  $\text{m}^{-2}$  in the baseline period as compared to  $1.8$  mol photons  $\text{m}^{-2} \text{day}^{-1}$  during dredging. NTU data can be converted to SSCs (as  $\text{mg L}^{-1}$ ) by applying site-specific algorithms (conversion factors) based on gravimetrically determined total suspended solid levels versus nephelometer readings. For the Barrow Island project, the conversion factor calculated this way



**Fig. 10.** Changes in turbidity and light before and during a major capital dredging programme where (A) Mean instantaneous NTU (every 10 min) and (B) Daily light integrals (DLI,  $\text{mol m}^{-2} \text{day}^{-1}$ ) during the Barrow Island project at 3 sites closest to the dredging (see Fig. 6A) in the pre-dredging (baseline) or during the dredging phase. Data are the mean of three sites close to the dredging (7–9 m deep and  $<100\text{--}500$  m away). Figures on the right hand side show the different percentiles over different running mean periods (from 1 h to 30 days) before (dashed lines) and during (solid lines) the dredging programme.

was 1.3–1.6 and average nephelometrically-derived SSCs over the baseline period were  $2.3 \text{ mg L}^{-1}$  (marginally above the  $\sim 1.5 \text{ mg L}^{-1}$  precision of the instruments) and  $7.1 \text{ mg L}^{-1}$  over the dredging project. The running means/percentile analysis was also conducted for the baseline and dredging seawater quality monitoring sites located from 0.2–15 km away from the principle dredging activity (the creation of the turning basins) during the Barrow Island project at sites for both the baseline and dredging periods and expressed as the  $P_{95}$  (Fig. 11). The turbidity patterns show a rapid initial decay in turbidity with increasing distance and effects on seawater quality could be observed up to 15–20 km from the dredging.

### 5.3. Light attenuation and light quality

Approximately 80 vertically-resolved downwelling planar irradiance light profiles were measured during the Cape Lambert dredging project using a Hydrolab-2 (Hydro-Optics, Biology, & Instrumentation Laboratories, USA). This radiometer provided irradiance measurement values at sub-nanometre spectral spacing. For each vertical profile the irradiance just-below surface  $E_d(0, \lambda)$  and the light attenuation coefficient  $K_d, \lambda$  were determined using the Beer-Lambert law. A general linear relationship between  $K_d, \lambda$  and surface SSC measurements was established for the dredge plumes encountered during sampling. This linear relationship, applied to all wavelengths between 350 and 850 nm, yielded a mass specific spectral attenuation coefficient for SSC, and an offset closely resembling the attenuation spectrum of seawater itself. Additionally, a relationship between clear sky extrapolated just-below surface incident irradiance  $E_d(0, \lambda)$  spectra and the solar zenith angle was determined.

Using these empirical relationships a downwelling irradiance spectrum was simulated for a given sun angle and water column averaged SSC concentration and PAR values were calculated from the full-spectrum measurement. Although empirically tied to the conditions found during the Cape Lambert dredging campaign, the model outputs demonstrated the general and spectral trends encountered in a dredge plume due to increased SSCs (Fig. 12A, B). This included a decrease in light with depth and increasing SSCs and a shift to more yellow light at high SSCs. For reference purposes, a vertically integrated  $10 \text{ mg L}^{-1}$  suspension of silt-sized sediments reduced light levels to  $\sim 11 \mu\text{mol photons m}^{-2} \text{ s}^{-1}$  at 10 m depth and a  $30 \text{ mg L}^{-1}$  SSC reduced the light to the same values by  $\sim 4 \text{ m}$  depth (Fig. 12A). At 5 m depth, the light spectrum under a dredging plume shifted almost entirely to yellow/green wavelengths at

concentrations of  $10 \text{ mg L}^{-1}$  and higher (Fig. 12B). These calculations were simulated for a clear (cloud-free) sky at solar noon, and during cloudy days at lower azimuth angles and at different sea states, underwater light quality and quantity would be substantially lower.

## 6. Experimental studies of the effect of sediments on corals

Laboratory-based studies examining the effects of sediments on corals can be grouped into experiments examining the effects in suspension and those examining the response of corals to a downward flux of particles. The latter group includes a range of burial and sediment covering experiments. As discussed further below, it is not clear if sediments were kept in suspension in the former group and the division between the tables is subjective.

### 6.1. Suspended-sediments

Some of the earliest studies of the effects of suspended-sediments were associated with understanding the sub-lethal and lethal toxicity of drilling muds and fluids on corals (Kendall et al., 1983; Szmant-Froelich et al., 1981; Thompson, 1980; Thompson et al., 1980) and several short-term feeding experiments investigating the effects of particulate matter concentrations on ingestion of corals (Table 1). However, the majority of the studies have been conducted over longer terms (weeks to months) and associated with examining the effects of sediments on corals to gain a better understand the effects of river runoff and/or dredging (Table 1). Whilst some studies have been conducted in situ using sediments introduced to small chambers held on the reef (i.e. Thompson et al. (1980) and Kendall et al. (1983)), most experiments were conducted in aquarium systems associated with marine research facilities. These ex situ studies have been conducted using either artificial lights (fluorescent and metal halides lamps) or under natural sunlight ( $< 300 \mu\text{mol photons m}^{-2} \text{ s}^{-1}$ ) using neutral density shade cloth to manipulate irradiance.

Sediments used in these studies have been calcium carbonate sands from the reefs where the corals were collected, sediments collected by filtering or back flushing filters, marl (a naturally occurring calcite), and kaolin clay (Table 1). The particle size distributions in these studies have not always been quantified and range from fine sands (i.e. Rice and Hunter 1992; Sofonia and Anthony, 2008), to fine silts ( $< 20 \mu\text{m}$ , Flores et al. (2012)). A combination of water pumps and aeration have typically been used to keep sediments in suspension.

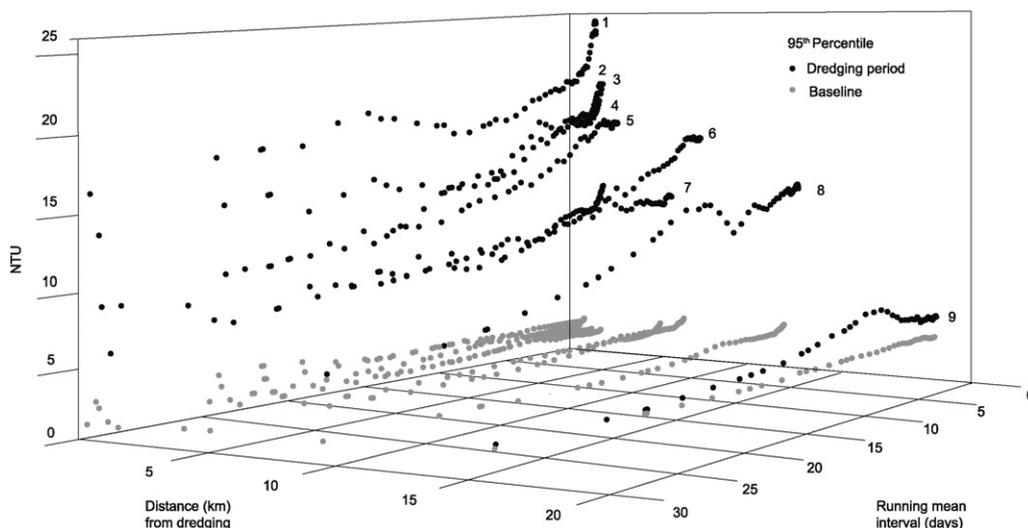
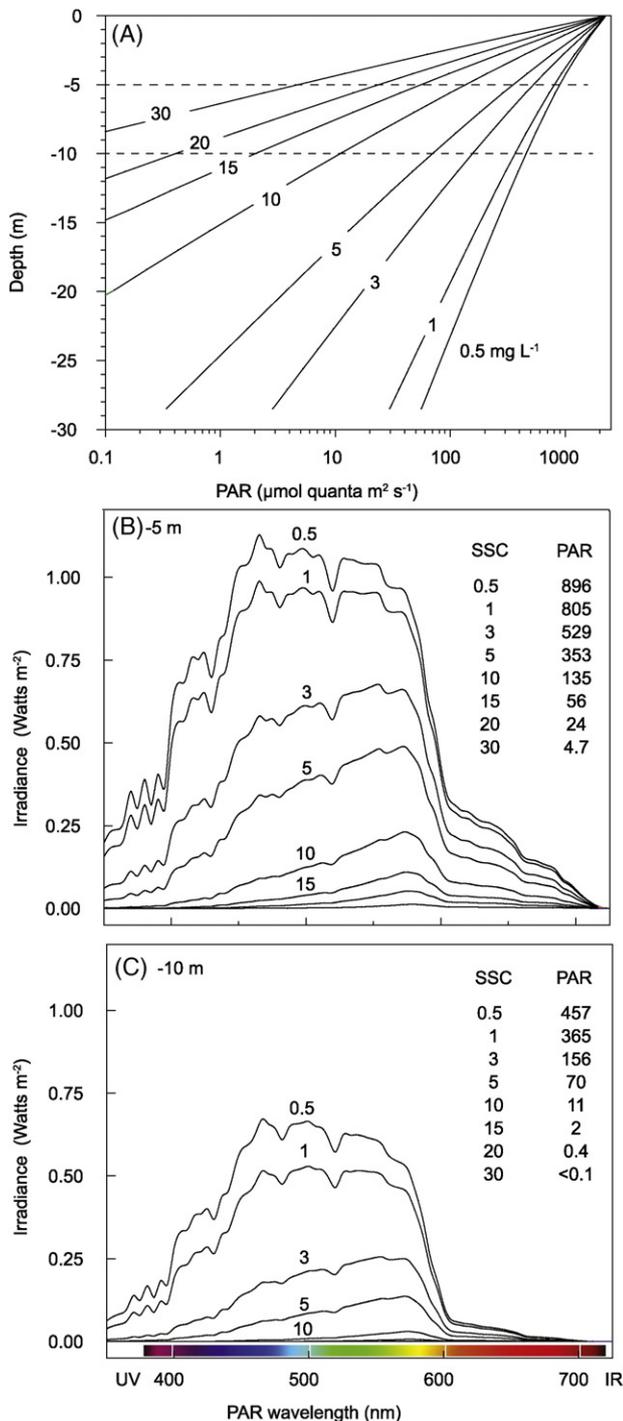


Fig. 11. Turbidity (NTU) values at different distances from dredging (from 200 m to 15 km away) and over different running mean time periods (from 1 h to 30 days). Shown are the 95th percentile values (see (1)) in the baseline period (before dredging, grey symbols) and during the Barrow Island dredging project (black symbols, see Fig. 6A).



**Fig. 12.** Simulated effects of suspended sediment on light quantity and quality based on empirical data collected during the Cape Lambert dredging project. (A) Simulated vertical PAR ( $\mu\text{mol photons m}^{-2} \text{s}^{-1}$ ) profiles for eight SSCs from 0.5–30  $\text{mg L}^{-1}$  (B and C) Simulated downwelling irradiance spectra at 5 and 10 m depth with varying SSCs from 0.5–30  $\text{mg L}^{-1}$  and associated PAR as  $\mu\text{mol photons m}^{-2} \text{s}^{-1}$ . The incident downwelling irradiance spectrum for all figures is typical of a clear sky at tropical noon, with an initial (underwater, 0 m depth) PAR value of  $\sim 1530 \mu\text{mol photons m}^{-2} \text{s}^{-1}$ .

## 6.2. Sediment deposition

The sediments used in deposition studies have typically been collected from local reefs or inter-reefal area and primarily composed of biogenic calcium carbonate or more terrestrial, siliciclastic sediments collected from river mouths (i.e. *Loiola et al., 2013*) (Table 2). One study used terrestrial quartz/granite beach sand (*Peters and Pilson,*

1985) and several studies have used silicon carbide (carborundum) for experiments on smaller particle sizes (e.g. *Bak and Elgershuizen, 1976*; *Stafford-Smith, 1993*; *Browne et al., 2014*) and limestone (*Logan, 1988*). Most studies employed a screening process to remove coarse and fine material and although PSDs of the sediments have only been quantified in a few studies, most have used sands, a few have used silts, and only the more recent studies have begun using fine, silt sized fractions, including the studies by *Weber et al. (2006)*, *Flores et al. (2012)* and *Weber et al. (2012)*.

The application methods have varied widely from manual application of sediments via a tube (i.e. *Bak and Elgershuizen, 1976*), funnel (*Riegl, 1995*), or pipette (*Logan, 1988*), a syringe (*Gleason, 1998*) or manually applied by unspecified techniques (*Schuhmacher, 1977*; *Lasker, 1980*; *Stafford-Smith, 1993*; *Sofonia and Anthony, 2008*; *Piniak, 2007*). In some of the studies, sediments were applied by creating initially high SSCs, which were then allowed to settle out of suspension on the corals (*Philipp and Fabricius, 2003*; *Weber et al., 2012*; *Weber et al., 2006*). For longer term exposures various sediment resuspension apparatus have used re-circulatory airlift systems (*Browne et al., 2014*; *Riegl and Branch, 1995*; *Todd et al., 2004*).

The majority of studies reported nominal sedimentation rates calculated by weight of sediment added and the surface area of the colonies. In some studies, it is not clear how sedimentation rates were calculated (*Lirman et al., 2008*) and in a few studies, sedimentation rate was measured using small sediment traps or plastic squares placed within the experimental containers (*Browne et al., 2015*; *Flores et al., 2012*; *Peters and Pilson, 1985*; *Sofonia and Anthony, 2008*; *Todd et al., 2004*).

## 7. Discussion

Our present day understanding of the effects of sediments on corals comes from observations and the results of many laboratory- and field-based manipulative studies in a body of literature generated primarily over the last 30 years. It has been reviewed many times (*Erfteimeijer et al., 2012*; *Fabricius, 2005*; *Foster et al., 2010*; *Jones et al., 2015b*; *Rogers, 1990*), and it seems reasonable to assume that results from these studies can be used for impact prediction purposes and deriving water quality values for managing dredging programmes near coral reefs. However, it is noticeable that seawater quality conditions during dredging programmes have never been quantified in detail (but see *Jones et al., 2015a*), and using information from several of the large scale dredging programmes close to coral reefs in recent years, the in situ seawater quality conditions differ in many regards from the experimental conditions used in past manipulative studies. In their review of principles of sound ecotoxicology and risk assessment, *Harris et al. (2014)* emphasize the critical importance of defining and testing realistic and environmentally relevant exposure scenarios and to comprehensively justify those exposure conditions (*Harris et al., 2014*).

The conceptual model developed in this review identified key cause-effect pathways affecting corals are light attenuation affecting photosynthesis (autotrophy), high SSCs affecting feeding and cleaning processes (heterotrophy), and sediment covering restricting solute exchange (smothering). These are well-known cause-effect pathways for the effects of sediments on corals (see *Rogers, 1990*), but from the conceptual model it is clear the proximal stressors are highly interlinked, with some connected along causal pathways and some acting alone or in combination. As such, the most relevant parameter(s) may change according to dredging activities, sea-state, distance from the dredge etc. Homer's sea-god Proteus was known for his ability to transform, i.e. to change shape and form. The 'protean' nature of suspended-sediments makes it difficult to identify which is the most relevant or important pressure parameter(s) at any given time, to identify cause-effect pathway(s), and importantly, to establish dose-response relationships. For example, if smothering of the coral's surface occurs, any biological effects which subsequently occur cannot be related to turbidity measured in the overlying water column. This has important

**Table 1**  
Studies examining the effects of suspended-sediments on corals. Experiments were conducted either under diel cycles or artificial lights from fluorescent and or metal halide lights. Under artificial lighting, the light intensities are expressed in  $\mu\text{mol photons m}^{-2} \text{s}^{-1}$  and Daily Light Integral ( $\text{mol photon m day}^{-1}$ ) calculated from the photoperiod. ns = not specified, SPM = suspended particulate matter.

Study	Species	Sediment type	PSD ( $\mu\text{m}$ )	Test type	Time (day)	SSCs ( $\text{mg L}^{-1}$ )	L:D cycle Light source	PAR (max $\mu\text{mol m}^{-2} \text{s}^{-1}$ ) DLI ( $\text{mol m}^{-2} \text{day}^{-1}$ )
Rice and Hunter (1992)	7 species of Atlantic corals	Offshore reef sediments	<500	33 min turnover	10–20	4 conc. 49–199	14:10 h Source: ns	PAR 2–2.5, DLI >0.13
Experimental outcome: No effects on survival in any treatments and no consistent patterns of a decline in growth rate								
Telesnicki and Goldberg (1995)	<i>D. stokesii</i> <i>M. meandrites</i>	Silt sized marlstone	<63	Partial water changes every 4 days	7–21	1–2, 7–9, 14–16, 28–30 NTUs	14:10 h Metal halide lamps	1–2 NTUs: PAR ~68, DLI 3.4 7–9 NTUs: PAR ~64, DLI 3.2 14–16 NTUs: PAR ~62, DLI 3.1 28–30 NTUs: PAR ~56, DLI 2.8
Experimental outcome: Effects on P/R ratios at 14–16 and 28–30 NTUs compared to controls. Prolonged exposure to 28–30 NTUs caused lesions in both species. NTU of 29–30 NTUs was equivalent to a Marl concentration on 55 $\text{mg L}^{-1}$ .								
Anthony and Fabricius (2000)	<i>G. retiformis</i> <i>P. cylindrica</i>	Collected by back-flushing a sand filter	ns	33 min turnover	56	<2 ~4 ~16	Natural cycle Sunlight + neutral density shade cloth	<2: PAR 600, DLI 8.1–12.8 ~4: PAR 140, DLI 2.5–4 ~16: PAR 140, DLI 2.5–4
Experimental outcome: Skeletal growth rates negatively affected by shading but not by the SPM treatments								
Anthony (2000)	<i>P. damicornis</i> <i>A. millepora</i>	Filtering of water from the reef	ns	Static	1 h	1, 4, 8, 16 and 30	Natural cycle Sunlight + neutral density shade cloth	PAR equivalent to 3–5 m depth across all treatments
Experimental outcome: Experimental outcome: Reduced feeding rates in corals exposed to 30 $\text{mg L}^{-1}$								
Sofonia and Anthony (2008)	<i>Turbinaria mesenterina</i>	Collected from an inshore turbid reef	60–120	Flow through with repeat dosing	34	<1, 16, 100 $\text{mg cm}^{-2}$	10:14 h Metal halide lamps	PAR ~300, DLI 11
Experimental outcome: No effect of the highest sediment load, regardless of flow conditions								
Flores et al. (2012)	<i>A. millepora</i> <i>M. aequituberculata</i>	Reef sediment ground using a ceramic mill	Mean 6	1.3 day turnover	84	1, 3, 10, 30, 100	12:12 h Fluorescent lights	1 $\text{mg L}^{-1}$ : PAR 200, DLI 8.3 30 $\text{mg L}^{-1}$ : PAR 184, DLI 8 100 $\text{mg L}^{-1}$ : PAR 177, DLI 7.6
Experimental outcome: For branching <i>A. millepora</i> some partial mortality occurred in the 100 $\text{mg L}^{-1}$ treatment (LOEC) but no mortality was noted at 30 $\text{mg L}^{-1}$ (NOEC). For <i>M. aequituberculata</i> (horizontal growth form) partial mortality was noted at 3 $\text{mg L}^{-1}$ and above, but related to settlement of sediment on the surface.								
Cooper and Fabricius (2012)	<i>Porites</i> sp	Collected from an inshore reef	<63	Continuous flow	56	2 and 20 under 2 shade treatments	12:12 h Metal halide lamps + neutral density shade cloth	1–2 $\text{mg L}^{-1}$ : 835, DLI 37 20 $\text{mg L}^{-1}$ : PAR 598, DLI 26 1–2 $\text{mg L}^{-1}$ : PAR 56, DLI 2.4 20 $\text{mg L}^{-1}$ : PAR 32, DLI 1.4
Experimental outcome no mortality in corals, change in colour brightness in corals exposed to SPM								
Browne et al. (2015)	<i>Merulina ampliata</i> , <i>Pachyseris speciosa</i> , <i>Platygyra sinensis</i>	Silicon carbide	1–300 Mean 60	Flow through	28	1 50–100 100–250	10:14 h High Output aquarium bulbs	1 $\text{mg L}^{-1}$ : PAR140, DLI 5 50–100 $\text{mg L}^{-1}$ : PAR 110, DLI 4 100–250 $\text{mg L}^{-1}$ : PAR 80, DLI 2.9

**Table 2**  
Studies examining the effects of settling sediments on corals.

Study	Species	Sediment	PSDs ( $\mu\text{m}$ )	Application methods	Application rate ( $\text{mg cm}^{-2} \text{ day}^{-1}$ )	Experimental outcome
Bak and Elgershuizen (1976)	19 Caribbean reef corals	Reef sand and carborundum powder	100–3000 mean = 1200	0.75, 1.5 and 3.5 g sand and 0.75 g carborundum delivered by a tube	100–425 (sand) and 100 (carborundum)	Sand cleared more efficiently than carborundum powder and coral surfaces were typically clean by 24 h in up to 3000 $\text{mg cm}^{-2}$ treatments
Schuhmacher (1974)	14 species from Gulf of Aqaba, Red Sea	Carbonate and silicate sand	>180 < 1.4 mm	Sand dropped on corals to cover the surface by a layer of 1 mm	Could not be determined	Most species covered with 1 mm of sand were still partially covered after 72 h
Lasker (1980)	<i>Montastrea cavernosa</i>	Collected from a reef, wet sieved and 2 fractions used	60–250 (fine) 500–1000 (coarse)	20 $\text{cm}^3$ of sediment deposited on each colony	18.5 (fine) 74.4 (coarse)	Approximately half to two thirds of the sediment removed from the surfaces within 8 h with most clearance occurring in the first 2 h
Peters and Pilson (1985)	<i>Astrangia danae</i>	Quartz/granite beach sand passed through <500 $\mu\text{m}$ sieve and combusted at 550 °C	62–500 mean = ~200	Sand sprinkled on the surface of the corals which were examined for 8 h	200 applied daily for 4 weeks then 3 times a day for 2 weeks	Sand rapidly (within 1–2 h) cleaned from the surface with little remaining after 24 h. No effects at the 200 $\text{mg cm}^{-2} \text{ day}^{-1}$ treatment but some cellular damage at the 600 $\text{mg cm}^{-2} \text{ day}^{-1}$ treatment
Logan (1988)	<i>Scolymia cubensis</i>	Fine, medium and coarse	>62–2000	3 g spread with a pipette over the corals' oral disk	Cannot be determined but likely to be >100 s	10–60% removed within 24 h in horizontally orientated corals, with fine sediments removed quicker than coarse sediment. Sediment clearance was 90% complete within 24 h if corals were inclined at 75°
Hodgson (1990)	<i>Oxypora glabra</i> , <i>Montipora verrucosa</i> , <i>Porites lobata</i> , <i>Pocillopora meandrina</i>	Fine, freshwater washed and dried marine sediment	'Fine' sediment	50 or 300 g of sediment stirred vigorously 1–2 $\times$ day	30–40 for 7–10 day	Sediments accumulated on flat and concave parts resulting in bleaching then tissue necrosis, and sediments overlying tissues changed to a greyish-black
Stafford-Smith (1993)	22 species of Australian corals	~70% carbonate & 30% quartz obtained from reefs	63–250 500–1000	5 g of sediment spread evenly over a 5 $\times$ 5 cm surface of the coral	200 and 200 for 6 weeks	Fine sediments were rejected more efficiently than coarse ones, with branching species clearing sediment loads within a few hours. Clearance of 50–100% of surface occurred within 24 h, with some less tolerant species failing to completely remove sediments after a few days. Bleaching and mortality was observed in some species within a few days, but other species required longer periods of sediment covering for mortality to occur
Riegl (1995)	8 species of corals from South Africa	Locally collected non-reef sand	v. fine (40–125) fine (125–250) coarse (>500) sand	Deposited through a funnel and a constant flow of fine sand through a re-circulatory system	200 for 8 h 200 for 6 weeks (continual coverage)	No difference between sediment type and 50–100% clearance of sediment in 8 h with no differences between species in the ability to shift different grain sizes. Mortality noted in 3 species after 15 days continuous exposure to 200 $\text{mg cm}^{-2} \text{ day}^{-1}$
Riegl and Branch (1995)	<i>Favia fava</i> , <i>Favites pentagona</i> , <i>Platygyra daedalea</i> , <i>Gyrosmlia interrupta</i>	Coarse reef sand combusted at 350 °C	46.1% 250–500 53.4% 125–250	14.2 g applied to a chamber containing a coral	200	Reduction in P/R ratios at all concentrations

Gleason (1998)	<i>Porites astreoides</i>	Intertidal beach sand Combusted at 450 °C	Mean = 253 70% <10, 25% 10–50 3% > 50–100 2% >100	Applied with a syringe	5–6 applied twice a day for four days	Sediment built up on the corals' surface resulting in many colonies exhibiting  Bleaching and early signs of necrosis in colonies exposed to 151 mg cm <sup>-2</sup> for 12–18 h and necrosis in corals covered from 24–36 h. Effects were confined to areas that were sediment covered  Some minor bleaching was observed, no mortality occurred and no obvious morphological changes  Sands were removed more efficiently than silts and nutrient poor sediments removed more efficiently than nutrient enriched ones. Anoxia develop under nutrient enriched silty sediments even under short term (days) exposure  Reduction in quantum yield in corals smothered in sediments for 30 h or more, with harbour mud having a greater effect than beach sand
Philipp and Fabricius (2003)	<i>Montipora peltiformis</i>	Collected from 3 m water depth from a local harbour	Fine, muddy sediment screened to <1000	Unspecified amount of sediment was placed into a 1000 L tanks and allowed to settle over 6 h	Average 151 (Range 79–234)	
Todd et al. (2004)	<i>Diploastrea heliopora</i> <i>Favia speciosa</i>	90%:10% quartz: carbonate	5% coarse, 45% medium, 45% fine and 5% very fine sand	Unspecified amount of sediment was resuspended for 1–5 min four times per day.	Average of 20 for 4 months	
Weber et al. (2006)	<i>Montipora peltiformis</i>	Collected from 2 rivers estuaries and 3 reefs, and aragonite dust from <i>Porites</i> skeletons	<63, fine sand (63–250) medium (250–500)	Unspecified amount of sediment was placed in 60 L containers and allowed to settle out over a 2 h period	Nominal values of 33, 66, 100, 133, 160	
Piniak (2007)	<i>P. lobata</i> and <i>M. capitata</i>	Carbonate beach sand and siliciclastic harbour mud	Sand (99%, 63–2000), mud (55%, 63–2000 and 23% < 0.62)	200–250 g of sand or mud spread evenly over the surface of the corals	<i>M. capitata</i> = 2200 (mud), 2,800 (sand), <i>P. lobata</i> = 1,500 mg (mud), 1,600 (sand)	
Sofonia and Anthony (2008)	<i>Turbinaria mesenterina</i>	Sediment (sandy silt) collected from a turbid reef environment	60–120	Fine sediment was applied manually 3 × per day for 34 days	<1, 16 and 100 each day	No effects observed
Lirman et al. (2008)	<i>Porites astreoides</i> <i>Siderastrea siderea</i>	Surficial sediments collected from a local reef	Sand with mean PSD of 176	1000 mg L <sup>-1</sup> sediment added daily to 5 L containers	53 per day for 3 weeks	Reduced growth rate in corals exposed to sand deposition for 3 weeks and involving a temporary covering of sediment and temporary reduction in light of 95% for 6 h each day  No mortality in <i>A. millepora</i> exposed to 30 mg L <sup>-1</sup> for 4 weeks and some partial and whole colony mortality at 100 mg L <sup>-1</sup> . Partial mortality in <i>M. aequituberculata</i> occurred as to as low as 3 mg L <sup>-1</sup> and 60% mortality occurred at 100 mg L <sup>-1</sup> . Mortality was directly related to sediment smothering which varied considerably between morphology the branching and plate like mortality
Flores et al. (2012)	<i>Acropora millepora</i> <i>Montipora aequituberculata</i>	Sediments were collected from an offshore reef	95% <20 µm mean size of <6 µm	Added to 30 L containers, and kept in suspension by pumps to give 1, 3, 10, 30, 100 mg L <sup>-1</sup> solutions	<1 to 83	Rapid (15–18 h) microbially mediated anoxia and tissue necrosis in corals covered in a 2–3 mm thick layer of organically enriched sediments
Weber et al. (2012)	<i>Montipora peltiformis</i>	Organic rich sediment collected from 5–10 m on a fringing reef of GBR	<63 µm mean = 15 µm	Sediment added to 60 L containers and held static for several hours to coat corals in a layer of sediment	66 = 2.1–2.6 mm thick	
Loiola et al. (2013)	<i>Mussismilia braziliensis</i>	Muddy, 91% siliciclastic sediment from a river channel	Fine silt	Sediment was added to 4 L containers and stirred for 2 min each day to resuspend the sediments	Unspecified amount of sediment allowed to settle over 24 h to produce 7 sedimentation rates of 0, 10, 50, 199, 250, 350 and 450	No necrosis in corals exposed to up to 150 mg cm <sup>-2</sup> day <sup>-1</sup> of organic free sediments for 45 days or up to 400 for 5 days
Browne et al. (2015)	<i>Merulina ampliata</i> , <i>Pachyseris speciosa</i> , <i>Platygyra sinensis</i>	Carborundum	1–300 µm median 60 µm	Corals exposed to intermittent bursts of sedimentation	4 week exposure to 25–65 and 4 week recovery period	No effects ( <i>Platygyra sinensis</i> ) <5% partial mortality ( <i>Merulina ampliata</i> ) or <15% mortality ( <i>Pachyseris speciosa</i> )

implications and potentially confuses and confounds laboratory and field experiments and the associated conclusions.

Summarizing the detailed seawater quality information in this study from Figs 7–11 (and also Fisher et al., in press; Jones et al. 2015a), in the near-field environment sediments released into the water column and moving out of the immediate dredging area (to create the far-field plume) are primarily silt- and clay-sized. Dredging increases the frequency of extreme values and alters the intensity, duration and frequency of the turbidity events over background levels. Upper percentile values (e.g.  $>P_{95}$ ) of seawater quality parameters can be highly elevated over short periods i.e. increasing by 2–3 orders of magnitude over a day and exceeding hundreds of  $\text{mg L}^{-1}$  over a period of hours. Over longer periods (days), SSCs  $P_{95}$  are in the tens of  $\text{mg L}^{-1}$  and typically less than  $10 \text{ mg L}^{-1}$  over a period of weeks and months. Scattering and light attenuation by the suspended-sediments occurs rapidly in the water column, with the shallow reef environment routinely experiencing semi-dark, caliginous, or 'twilight' periods, and frequently complete loss of light at moderately elevated SSCs. However, a more common feature was extended periods, i.e. days to weeks, of low light. Different wavelengths are preferentially attenuated underneath plumes, with more immediate loss of red and blue light and a shift to less photosynthetically useable yellow-green light. The impacts of dredging on sedimentation and the problems associated with measuring sediment deposition and the significance of quantifying sediment deposition for understanding coral mortality in dredging programmes are discussed further below.

#### 7.1. Experimental studies of the effects of suspended-sediment and light attenuation on corals

It is not clear from the description of many of the laboratory based studies whether sediments were truly kept in suspension throughout the study, especially those with larger particle sizes (i.e. Rice and Hunter 1992) and higher sediment concentrations (Browne et al., 2015; Sofonia and Anthony, 2008). Many of the laboratory-based studies examining the effects of suspended-sediment have not specified the particle sizes and therefore it is difficult to ascribe any observed effects to SSCs or to sediment covering. All studies were conducted in shallow containers and it is conspicuous that many studies have not compensated for the fact that light attenuation is exponential, and likely to be quite small in such a shallow water depth even at higher concentrations. Flores et al. (2012) measured a  $<10\%$  reduction in PAR in 15 cm deep experimental containers caused by a  $30 \text{ mg L}^{-1}$  SSC of fine silt, whereas on a reef at solar noon and on a sunny day, nearly all light would be attenuated by 5 m under an equivalent concentration (Figs. 9, 10 and 12). Some studies have recognised this issue and compensated for this depth effect by reducing light levels with neutral density shade-cloth at higher sediment concentrations, or partially compensated for the effect by reducing light uniformly across several different sediment concentrations (Anthony, 2000; Anthony and Fabricius, 2000; Cooper and Fabricius, 2012). Other studies have not addressed the issue at all (Browne et al., 2015; Flores et al., 2012; Hodgson, 1990; Riegl, 1995; Sofonia and Anthony, 2008; Telesnicki and Goldberg, 1995; Thompson et al., 1980). This can make the results misleading for hazard assessment purposes, i.e. corals could be living through SSC treatments in laboratory-based experiments which in situ could have more profound effects because of the associated light reduction. In the experiments of Browne et al. (2015), PAR levels in the shallow containers during high sediment pulses ( $50$ – $100$  and  $100$ – $250 \text{ mg L}^{-1}$ ) were reduced from  $140 \mu\text{mol photons m}^{-2} \text{ s}^{-1}$  to  $100$  and  $\sim 70 \mu\text{mol photons m}^{-2} \text{ s}^{-1}$ , respectively. Because there was no additional shading to reduce the light levels to the levels corals would experience in situ under similar suspended-sediment concentrations, the corals lived through the 4 week exposure with either no effects (*Platygyra sinensis*)  $<5\%$  partial mortality (*Merulina ampliata*) or  $<15\%$  mortality (*Pachyseris speciosa*). Survival of corals in situ under more appropriate light regimes (i.e.

most probably full light extinction – see Fig. 12A) over a similar extended time period, could have resulted in a different outcome given the physiological effects known to occur in corals in darkness (DeSalvo et al., 2012; Rogers, 1979; Yonge and Nicholls, 1931).

Compensating for the depth effect using neutral density shade cloth over containers will not, however, correct for spectral changes which may occur underneath dredge plumes. The underwater light field is modified by the spectral-dependent absorption and scattering properties of the seawater itself involving absorption of the red-infrared-spectral region, and also by phytoplankton, humic substances (gelbstoff) and especially particulate matter (Jerlov, 1976; Kirk, 1994; Kirk, 1985). The hyperspectral data showed a reduction in blue and red wavelengths and a clear shift to yellow-green conditions underneath a dredge plume. This region is outside of the major absorption peak of photopigments, meaning that light is poorly absorbed by the corals in this part of the spectrum and it is relatively inefficient at driving photosynthesis (Halldal, 1968; Szabó et al., 2014). This is a significant issue with monitoring programmes and for deriving in situ dose-response relationships from laboratory experiments, because the types of light sensors commonly used in monitoring programmes integrate across the PAR wavelengths and do not account for any spectral changes. That is, the reported light levels in 9, 10, and 12 could be misleading as to what corals can actually use for photosynthesis, i.e. the difference between photosynthetically useable radiation (PUR) (Morel, 1978) and photosynthetically active radiation (PAR) (Tyler, 1966). A number of recent studies have begun investigating the effects of light quality on coral physiology (Wangpraseurt et al., 2012; Wangpraseurt et al., 2014) made easier by recent advances in lighting technology (Wijgerde et al., 2012; Yeh et al., 2014), and spectral changes should be considered in future studies if results from laboratory based studies examining SSCs and light reduction are to be extrapolated to the field.

The results from those experiments where there was no light compensation have inadvertently provided evidence to suggest corals can actually survive quite high SSCs as long as the light is sufficient and there is no sediment accumulation on the surfaces. For example, there was no mortality in *A. millepora* exposed to  $\text{mg L}^{-1}$  for 30 days (Flores et al., 2012), or in *Porites* spp. exposed to  $20 \text{ mg L}^{-1}$  for 56 days (Cooper and Fabricius, 2012). Skeletal growth of *Goniastrea retiformis* and *Porites cylindrica* exposed to  $16 \text{ mg L}^{-1}$  for 2 months were also not different from ex situ or in situ controls (Anthony and Fabricius, 2000). These sediment loads (intensity  $\times$  duration) are high compared to natural resuspension events (Larcombe et al., 1995) and conditions which can occur close to a major capital dredging project (see Figs. 10, 11). By isolating SSCs as the primary variable, these studies have provided evidence that for adult corals, light availability and sediment covering are probably the most important cause-effect pathways associated with turbidity generating events in the short term.

#### 7.2. Experimental studies of the effects of sediment deposition on corals

There have been many studies examining the effects of sedimentation on corals but there are also some methodological issues which make interpretation of these studies difficult. Many studies have used silicon carbide as a sediment proxy (Browne et al., 2015; Junjie et al., 2014; Lui et al., 2012; Stafford-Smith and Ormond, 1992), which because of differences in specific gravity, sphericity and porosity and light scattering and absorbing properties makes generalization extremely difficult (Storlazzi et al., 2015). The use of black carborundum, which traces back to the feeding experiments of Yonge (1930), should be considered carefully if the intention is to use the results of such studies to make generalizations of the effects of sediments released from dredging. As significantly, it is noticeable that many studies have used sands ( $62$ – $2000 \mu\text{m}$ ) as opposed to the fine silts and clays which typify the near and far-field dredge plume. Smaller particles have much greater scattering and absorption properties and so light transmission and solute exchange will vary considerably depending on sediment type

(and colour) (Storlazzi et al., 2015). Weber et al. (2006) recorded light transmission of only ~0.1% through a 66 mg cm<sup>-2</sup> sediment layer on a coral in contrast to Riegl and Branch (1995), who recorded light transmission as high as 30% through a 200 mg cm<sup>-2</sup> sediment layer. The primary difference between the studies was that Weber et al. (2006) used silt (with a median grain size of ~10 µm) and Riegl and Branch (1995) used much coarser sand with a median grain size of 250 µm (~50% between 250 and 500 µm).

The difference between the silt and clay sized particles found in dredging plumes and the sands used in some of the laboratory experiments makes it difficult to interpret information and extrapolate to conditions during dredging. For example, in experiments designed to partition the effect of turbidity and settling particles on *Galaxea fascicularis* and *Goniopora somaliensis*, Junjie et al. (2014) acknowledged that settled sediment could create an additional barrier to light, but then discounted the potentially confounding effect on the high light availability reported by Riegl and Branch (1995) discussed above. Junjie et al. (2014) used substantially smaller grain size (10–300 µm with a median grain size of 60 µm), used black carborundum (see Storlazzi et al., 2015), and the possibility that light attenuation by the sediment layer was not a significant factor in this experiment study has not been discounted.

Why such coarse sediments have so routinely been used is not clear, but perhaps the reason is purely practical, as silts and clay cloud the seawater so much that the corals cannot be seen (see Bak and Elgershuizen, 1976; Todd et al., 2004). Settling velocities of sediments are related to density and proportional to the diameter squared according to Stoke's law. Assuming particles do not coagulate, flocculate, or grow and in the absence of vertical mixing, a fall velocity of 1.29 cm s<sup>-1</sup> for sand-sized (200 µm) particles (Storlazzi et al., 2011) should result in sediment at the surface reaching a 10 m deep seabed within <15 min. Settling velocities of silts are lower and they have the opportunity to move away from dredges onto nearby habitats. Far field plumes (km away from dredges) are likely to be made up of smaller particles still, similar to the fine silts and clays which only recently have been reported as dominating river plumes a few km from river mouths (Bainbridge et al., 2012).

A different and perhaps more fundamental problem than sediment type and particle size is the application rates (sedimentation rates) used in these studies. All the studies have used experimental sedimentation rates based on in situ measurements with sediment traps. The problem with traps is that they capture all particles including those which are just passing over the reef or that are only momentarily deposited (Field et al., 2012; Risk and Edinger, 2011; Storlazzi et al., 2011; Thomas and Ridd, 2004). Traps can even collect sediments that never actually settle and therefore can provide a sedimentation estimate (even in net erosional environments), i.e. they provide a 'pseudo-sedimentation' rate. Interpreting data from traps is more difficult in shallow, energetic environments that are characteristic of reefs, and these limitations do not include the effects of waves and wave orbital velocities which result in acceleration and deceleration of horizontally moving particles over the trap mouth during agitated sea states (Bothner et al., 2006; Storlazzi et al., 2011) (cf Fig. 1). Traps have been described as at best only providing a vague approximation (Storlazzi et al., 2011) of the amount and type of sediment that deposits, and have been suggested as being more apt to record information on suspended-sediment dynamics than provide any useful data on sedimentation (Buesseler et al., 2007; Gardner, 1980; Gardner et al., 1983; Storlazzi et al., 2011). These problems have been discussed and reviewed many times in different fields of marine research, see for example (Bothner et al., 2006; Browne et al., 2012; Buesseler et al., 2007; Butman et al., 1986; Jürg, 1996; Kozerski, 1994; Reynolds et al., 1980; Risk and Edinger, 2011; Storlazzi et al., 2011; Thomas and Ridd, 2004).

A number of studies have attempted to overcome the trapping artefact and address the question of what are typical sedimentation rates on

coral reefs. On Kenyan reefs, McClanahan and Obura (1997) noted that a factor of 3 is required to scale up sedimentation rates on flat tiles (which do not suffer resuspension limitation and deposition bias) to those measured with sediment traps (range 3–6 mg cm<sup>-2</sup> day<sup>-1</sup>). More recently, using circular, concrete-filled PVC hubs (SedPods), Field et al. (2012) measured net sedimentation rates of 0.3–0.6 mg<sup>-2</sup> day<sup>-1</sup> in Hanalei Bay, Hawaii, as opposed to sediment trap accumulation rates of 7–17 mg cm<sup>-2</sup> day<sup>-1</sup>. Using shallow trays, which also allow resuspension, Browne et al. (2012) estimated that net deposition rates in the high-turbidity nearshore coral settings on the inner shelf of the central Great Barrier Reef, averaged 3–7 mg cm<sup>-2</sup> day<sup>-1</sup> over the course of a year. These values were considerably lower than measured using traps in the same area i.e. a mean of 44 mg cm<sup>-2</sup> day<sup>-1</sup> with an upper value as high as 364 mg cm<sup>-2</sup> day<sup>-1</sup> (Mapstone et al., 1992). These more recent studies of deposition rates need to be contrasted with earlier suggestions – but based on the current understanding at the time – that sedimentation rates in excess of 200 mg cm<sup>-2</sup> day<sup>-1</sup> for periods of days to weeks are '...not uncommon on fringing reefs of the GBR...' (Stafford-Smith, 1993).

The problem with interpreting information from traps led Storlazzi et al. (2011) to conclude that prior research results in the literature need to be interpreted carefully and with recognition that there may be irregularities in the trapping technique or in the application to understanding coral reef processes. Within the context of developing thresholds for dredging projects, if sediment traps are overestimating sedimentation rates (through resuspension limitation and deposition bias), then replicating those rates conditions in laboratory studies may result in exposure scenarios that are unrepresentative of anything but extreme conditions. The experimental application rates are often in the high tens of mg cm<sup>-2</sup> day<sup>-1</sup>, commonly hundreds of mg cm<sup>-2</sup> day<sup>-1</sup>, and sometimes even g cm<sup>-2</sup> day<sup>-1</sup>. When these rates are applied under still or low flow conditions this can result in sediment deposits that are millimetres thick on the corals' surface. The deposition rate of 66 mg cm<sup>2</sup> day<sup>-1</sup> in the studies of Weber et al. (2012) resulted in a smothering of the coral tissues in a 2–3 mm thick layer of sediment, and this is likely to be less than the earlier deposition experiments of Philipp and Fabricius (2003) where deposition rates of up to ~230 mg cm<sup>2</sup> day<sup>-1</sup> were used. For comparative purposes, the visually significant turbid flood plumes on the Great Barrier Reef have concentrations of sediments which, if settled to the seabed, would produce a deposit of only 10 µm (Orpin et al., 2004). The question then becomes how representative are experiments that create mm thick deposits of sediment in less than a day, and do these reflect conditions that can occur naturally or only very extreme events such as cyclones or only conditions that occur very close to dredging.

Smothering is regularly observed in dredging projects and occurs because the sedimentation rate is likely to exceed any natural rates corals have experienced previously. This is because dredging can cause high SSCs in sea-states where ambient hydrodynamics cannot support the load (Fig. 3A–D) and the sediment 'overburden' rapidly falls out of suspension according to particle specific settling velocities. Smothering occurs when sediment cannot be cleared fast enough from the surface and sediment begins to accumulate over successive days producing the sort of images in Bak, 1978; Foster et al., 2010, and Fig. 3. In contrast, natural high SSCs produced by high wind events occur where wind-driven waves and tidal currents create conditions where wave orbital velocities are sufficient to keep some of the sediments in suspension and deposition subsequently occurs (after a settling lag) during quiescent periods and after further entrainment and dilution (Ogston et al., 2004). These are very different scenarios and sediment deposition is therefore different from other proximal stressors such as elevated SSCs and light reduction, as short term turbidity events and low light, twilight, periods are not uncommon in the marine environment associated with wind and wave events (Anthony and Larcombe, 2000; Jones, 2008; Storlazzi et al., 2009). For these proximal stressors it is likely to be the duration and frequency caused by dredging

that differs from natural conditions rather than the intensity as with sediment deposition. Corals are exposed to conditions (of low light and high SSCs) that they have experienced previously, in the short term, and are physiologically acclimated to these natural conditions. How they do this is via a range of strategies including photo-acclimatory changes, shifts from autotrophy to heterotrophy, and replenishment of energy reserves between turbidity events (Anthony, 2000, 2006; Anthony and Fabricius, 2000; Anthony and Hoegh-Guldberg, 2003; Anthony and Larcombe, 2000).

There are currently no suitable techniques for measuring low mg per cm<sup>2</sup> deposition events with sufficient resolution to be effective as a monitoring tool for dredging programmes (Field et al., 2012; Perkey and Wadman, 2013; Risk and Edinger, 2011; Thomas and Ridd, 2004), although use of upward pointing optical backscatter devices offer promise (Thomas and Ridd, 2005; Thomas et al., 2003). Once information becomes available on the range of sediment deposition rates (as mass per unit area or thickness deposition per day) over different time periods during natural conditions, or in the near- and far-field during dredging programmes, then this will allow contextualization of past studies and information that can be used for impact prediction purposes for sedimentation.

### 7.3. Seawater quality thresholds for coral reefs and future directions

Future ex situ studies need to clearly state what pressure parameter is being tested and recognise and eliminate the potential confounding effect of other parameters if values are to be proposed for threshold development. Studies should be conducted with locally collected reefal sediments rather than using sediment proxies such as carborundum. Studies to characterize the organic content of the sediments released into the water column by dredging (i.e. loss at the drag/cutter heads, by overflowing, or released by disposal at placement sites), and how it compares to surficial samples from the seabed before dredging are needed. Many studies have now shown the importance of the organic content of the sediments on coral sediment clearance and survivorship (Loiola et al., 2013; Weber et al., 2012; Weber et al., 2006). The use of low organic content or organic free sediments (Peters and Pilson, 1985; Riegl, 1995; Riegl and Branch, 1995), or proxies (Browne et al., 2014; Lui et al., 2012) has been suggested a starting point for understanding coral sediment shifting capabilities without additional variable of organic content, nutrient, and microbial content. Particle sizes should be measured and experiments examining the effects of sediment deposition in the near-field need to use up to coarse silt-sized sediments and finer fractions for far-field conditions. Contaminant levels need to be described if there is reasonable doubt they are of concern. PAR levels in tanks should be specified as maximum, as well as the daily light integral and recent advances in lighting technologies such as light emitting diodes (LEDs) means it is now possible to address spectral changes under plumes.

As Harris et al. (2014) warn in their review of principles of sound ecotoxicology, there is a danger associated with an incomplete understanding of exposure pathways and the use of conditions that are unrepresentative of the majority of situations typically encountered by wildlife. This statement seems applicable to studies on the effects of sediments on corals (see also Storlazzi et al., 2015; Storlazzi et al., 2011), as many experiments have been conducted without explicit justification of the exposure regimes especially with studies examining sedimentation. Harris et al. (2014) suggest authors should be open and honest about the context of their study to those conditions which have been measured (or predicted) in the real environment and the explanation of the exposure conditions for futures studies should be comprehensive. This will then enable judgements to be made of whether high suspended-sediment concentrations and the associated light reduction and sediment smothering at different distances from dredging is a hazard or a risk to underlying communities, and allow the development of

the water quality thresholds for impact prediction and monitoring purposes.

### Author contributions

Conceived the study RJ, PB-B, RF. Conducted the analyses, and developed the model RJ, P B-B RJ. Conducted the PSD and light quality and quantity measurements MS, WK. All authors contributed to the writing and approved the final review.

### Competing interests

The authors have declared no competing interests exist.

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