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Effects of dredging and dredging related activities on water quality: Impacts on coral mortality and threshold development

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WAMSI Dredging Science Node

Report 4

Project 4.9

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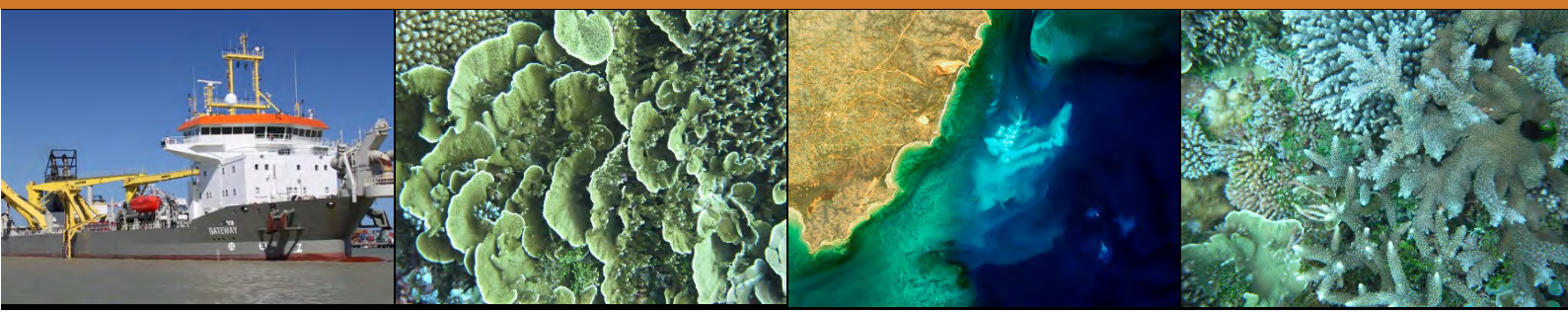

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WAMSI Dredging Science Node

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

Funding Sources

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

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

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

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Front cover images (L-R)

Image 1: Trailer 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 August 2010.

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

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Executive Summary

Dredging and related activities are common across nearshore marine environments, potentially threatening nearby ecosystems. Regulatory frameworks are essential for balancing environmental concerns against the need for port expansion and coastal development. In particular, thresholds that relate the physical pressures[1] to the biological response(s) and define exposure conditions above which effects will occur are a key tool used for minimising environmental impacts. Thresholds are required to predict zones of potential impact during the pre-dredging environmental impact assessment (EIA) phases, as well as during the operational phase as triggers for management actions. The process of deriving and practically applying such thresholds is a complex issue that relies heavily on a sound understanding of how ecosystems will respond to environmental stressors.

Guideline or threshold values can be developed from both *ex situ* laboratory-based (i.e. aquarium) studies and also from *in situ* field-based studies. The pros and cons of the two approaches are well known (see Theme 4.6 *Laboratory-based studies examining the effects of sediments on corals*) and guideline development is most robust when based on a combination of both types of studies and using physiological and ecological endpoints and field observations i.e. using a weight of evidence (WoE) approach. This combined approach is possible in this Theme because of data water quality and coral health datasets from four large scale dredging projects in the Pilbara which were made available to the Dredging Science Node for scientific study[2] [3] [4] [5]. The Barrow Island project coral health and water quality datasets are a particularly rich source of information.

The report is divided into three parts including (1) analysis of the Barrow Island coral health data set, providing qualitative and quantitative analysis of the various drivers of coral mortality (Supporting Publications and Reports - 1 & 2), (2) the derivation of water quality management thresholds (Supporting Publications and Reports - 3) (Fisher et al. 2018) and (3) a more detailed discussion of some of the broader complexities of operationalising thresholds, providing context for some of the many assumptions underlying the threshold derivation in (Fisher et al. 2018), the logic behind the theoretical approaches used (Supporting Publications and Reports – 4). The report finishes with a synthesis of the Theme 4 findings to develop an outline of how thresholds should be used operationally during the EIA phase and for active dredging management.

The Barrow Island coral health data set

Analysis of the mortality pathways during the Barrow Island provided many insights into the cause of dredging related mortality and how this varied among different coral types (Projects 4.9.1, 4.9.2).

Sediment deposition resulting in smothering of corals was one of the key cause-effect pathways associated with dredging near coral reefs, however these effects are much closer ('10% Effect Distance' (ED_{10}) for the % of colonies showing mucous and sediment covering of >5% at 3-3.3 km) than the observable changes in water quality conditions (ED_{10} at ~15-20 km), with more substantial impacts (e.g. ED_{50}) only occurring within less than one kilometer. The response of coral colonies to high sediment loads varied across coral taxa and growth forms, with smothering common on encrusting, foliose and certain massive morphologies, but never observed on branching species. In morphologies where smothering occurred, sediments were commonly seen accumulating in concave depressions or 'hollows', sometimes requiring considerable wave action for removal. Sediment smothering can cause tissue bleaching, but depending on the duration and extent of smothering, bleached areas

[1] A *pressure* is a physical, chemical or biological change that has the potential to cause environmental change. In terms of dredging activities the term is used to signify elevated SSCs, sediment deposition, or reduced light availability from increased turbidity.

[2] Pluto LNG Development, Burrup Peninsula: WA Environmental Protection Authority Bulletin 1259, Ministerial Statement No. 757

[3] Cape Lambert B project: WA Environmental Protection Authority Bulletin 1357, Ministerial Statement 840

[4] Gorgon Gas Development Barrow Island Nature Reserve: WA Environmental Protection Authority Bulletin 1221 Ministerial Statement No. 800

[5] Wheatstone (Onslow) Development - Gas Processing, Export Facilities and Infrastructure: WA Environmental Protection Authority Bulletin 1404 Ministerial Statement No. 873

may recover, or may result in necrosis and lesions (although even these sometimes show evidence of recovery).

A warm-water bleaching event occurred approximately mid-way through the Barrow Island dredging project. Overall observed mortality across sites was driven by a combination of the environmental pressures associated with dredging, as well as bleaching of corals through thermal stress, with both dredging related pressure metrics as well as temperature and bleaching being strong predictors of coral mortality (Project 4.9.1). Specific dredging related pressure metrics varied among taxa, with the branching *Acropora* and Pocilloporidae showing strong relationships with light loss and the massive Poritidae also heavily influenced by sedimentation (Fisher et al. In prep). Importantly, the suspended sediments themselves showed little direct influence on mortality, highlighting that, while SSCs are directly responsible for generating both low light and high sedimentation stress, they do not appear to be an important pressure in their own right, providing strong field support for the findings of Theme 4 laboratory studies (Bessell-Browne et al. 2017a). There is evidence that depending on the severity of their impact on benthic light and sedimentation, suspended sediments can have both negative and positive effects on corals during periods of thermal stress. Low to moderate reductions in light alleviate coral bleaching through shading, meaning lower overall mortality of coral (antagonistic cumulative impacts). Conversely, at high sediment loads any positive effect of reduced bleaching is outweighed by higher levels of mortality associated with sediment related stress, and actually results in higher overall levels of mortality of coral (synergistic cumulative impacts). The optimal light level for the antagonistic shading effects occurred at around 3.1 mol photons m^{-2} d^{-1} for the reefs at Barrow Island, which may prove to be a useful target for active mitigation efforts involving shading. In addition, the findings suggest that while there may be no reason to limit sediment generating activities (such as dredging) near coral reefs if the water quality conditions remain better than the transition from antagonistic to synergistic impacts (~1.2 mol photons m^{-2} d^{-1}), at higher sediment loads the cumulative impacts of suspended sediments and thermal stress may be much worse when they occur concurrently.

Because of the complexities associated with the thermal bleaching event at Barrow Island, it was decided that to derive *in situ* thresholds for managing future dredging projects (see below) analyses were based on the water quality and coral health only during the pre-bleaching phase from the start of dredging up to immediately prior to when bleaching impacts become evident (after day 203). This enabled thresholds for dredging related stressors to be derived without the confounding influence of other cumulative factors (Fisher et al. 2018, Project 4.9.3).

Estimating guideline values using the Barrow Island coral health data set

Thresholds were estimated using novel statistical approaches and mathematical tools to account for both the uncertainty in site level responses to dredging related pressures and the inherent statistical uncertainty associated with imperfect metrics of exposure (Fisher et al. 2018, Project 4.9.3). Bayesian statistical methods were used to factor out natural mortality (by differencing against mortality observed at reference locations) and calculate probabilities of non-zero mortality of corals across the first 203 days of the pre-bleaching dredging period. These probabilities were used to derive dose-response relationships against a broad range of water quality metrics capturing each of the 3 primary mortality pathways (turbidity as a surrogate for SSCs, loss of benthic light availability and sediment deposition) across 3 exposure types (running mean, percentage exceedance and consecutive exceedance) representing different elements of intensity (I), duration (D) and frequency (F).

In general, all water quality metrics examined were relatively strong predictors of the probability of non-zero coral mortality. In particular deposition based metrics tended to have the strongest relationships – but also pose significant issues because sediment deposition is inherently difficult to measure (Whinney et al. 2017a). Recommendation on how to circumvent this issue is discussed below in Section 3. Both turbidity and light are relatively easy to measure *in situ* and despite lower statistical power, were also explored as candidate threshold metrics.

The best metrics for the primary mortality pathways and 3 exposure types were used to formally derive management thresholds using modified receiver operating characteristic (ROC) curves. This method enabled

derivation of thresholds with explicit Type I and Type II errors rates, thus generating a matrix of candidate thresholds across a broad range of metrics reflecting different management objectives that can be applied to future dredging programs in equivalent environments (see Table 1, (Fisher et al. 2018) and Management Implications below). Thresholds reflecting aversion to a false sense of security in environmental protection were referred to as 'Strict' thresholds (synonymous with *possible effects)* and had low Type 2 error rates and high statistical power associated with them. 'Permissive' thresholds reflected aversion to the costs of false alarms (and are synonymous with *probable effects*) and had low Type 1 errors rates. The two types of thresholds varied substantially for all exposure metrics examined, highlighting the need to be mindful of both the inherent statistical error associated with thresholds (which is often overlooked), as well as the threshold's intended management objectives.

Complexities associated with developing and operationalising thresholds

Practical considerations for threshold implementation

NTU–SSC conversions - Water quality monitoring programs in recent capital dredging projects in NW WA have typically focused on measuring turbidity, using optical back scatter technologies (nephelometers), which can be used to provide proxy estimates of SSCs. The emphasis on SSCs is intuitive because it is associated with the sediment itself which is released into the water column by dredging, but importantly also because SSCs can be modelled during the pre-dredging EIA stage using coupled sediment transport and hydrodynamic models that predict sediment transport and fate. Analyses of the NTU-SSC relationships indicated considerable variability within and between studies and that greater attention must be taken to check and reconfirm the NTU–SSC conversion factors throughout dredging programs if nephelometers are used as proxy measurements of SSCs.

Data aggregation - Fine temporal scale data (i.e. 10 min/30 min) collected during dredging projects are often aggregated to coarser temporal scales. An analysis was conducted to show that how data is treated i.e. geometric versus absolute mean can influence statistical summaries and how data are handled, in the context of transformations and averaging, must be carefully considered and always clearly reported, particularly where running mean values are used as thresholds in management.

Cyclones - Depending on their size and proximity, cyclones can substantially affect water quality data (turbidity, PAR and sediment deposition) on a par with the effects of dredging. An analysis of water quality during the baseline phases of 4 dredging projects showed the P_{80} of the baseline turbidity could be as much as 60% higher if cyclones are included in the analyses.

Threshold derivation

Threshold durations - The process of threshold derivation is inherently complex as there are trade-offs between intensity, duration and frequency of dredging pressures and associated impacts on coral health, and it is important that guideline values incorporate a temporal component. Temporal scales of thresholds are not always clearly evident, nor is the relationship between duration and intensity explicitly articulated. Time components must to be built into water quality thresholds that explicitly allow the calculation of accumulated pressure encompassing both intensity and duration. These should be expressed in absolute terms (i.e. x mg L^{-1} for y days or over period y) as opposed to relative (to reference locations) guidelines (i.e. x mg L^{-1} for y days or over period y, over a background of z mg L^{-1}). Guidelines such as SSCs >x mg L^{-1} for less than 20% of the time are non-specific with respect to how far SSCs can exceed x mg L^{-1} or when the exceedance occur (consecutive days or intermittently over a longer period) both of which could have material consequences for corals. Several lines of evidence suggest that a two week time frame may often be a practicable and useful time scale for threshold durations, as this corresponds reasonably well with time scales of observing biological effects in experimental settings, there is evidence from the field data that 14 day running means provide strong relationships with coral mortality, and the fact that a period of a few weeks allows dredging proponents time to plan and enact adaptive management activities as well as having reasonable confidence in future weather and tidal patterns.

Use of baseline data for threshold development - There are numerous methods for deriving thresholds, which

can result in substantially different values. Methods that utilise baseline data to estimate thresholds, such as the ANZECC/ARMCANZ P50–P80 approach (ANZECC/ARMCANZ 2001) and the Intensity-Duration-Frequency [IDF] approach of MacArthur (McArthur et al. 2002) yielded far more conservative threshold values than those based on observed coral mortality (e.g. Fisher et al. 2018, Project 4.9.3). At most sites, thresholds based on realised coral mortality are greater than the 99th percentile of the baseline time series, even when the *strict* (indicative of possible coral mortality) thresholds were considered. Such high percentile values are statistically unstable, and extrapolation of thresholds across different locations using percentiles is not recommended.

Multiplier based threshold extrapolation - Another more general approach that could be used to estimate guideline values when no data are available on the sensitivity of benthic organisms, is to use a multiplier technique i.e. multiplying the baseline levels by a certain value. Multipliers to reach the *strict* (possible effects) and *permissive* (probable effects) across sites at Barrow Island were 2.2 and 3 times the mean baseline conditions for turbidity, and 1.4 and 1.6 times the mean baseline conditions for light stress respectively.

Spatial variation in background conditions – There is considerable variation among sites in the background turbidity and light regimes across the four most recent capital dredging locations in the Pilbara. We explored the extent which a multiplier approach might be used to extrapolate thresholds to locations with higher or lower background turbidity, and lighter or darker background benthic light conditions. Applying the mean (across sites) multipliers at Barrow Island can yield some very extreme threshold values at locations with very different background conditions, and this approach is not recommended without further validation that the theoretically derived thresholds are in fact appropriate.

Sediment deposition and coral health

Sedimentation has been identified as a key pathway causing mortality of corals during dredging, particularly for massive corals (Fisher et al. In prep, Jones et al. In prep), yet remains problematic as it is difficult to measure in the field. While new methodology has been developed for measuring ecologically relevant deposition *in situ* (Whinney et al. 2017b), this method has no existing historical data that can be used to derive thresholds. Until thresholds for reliable *in situ* deposition measurements have been established, in the interim turbidity may be used, as this can be correlated with a sediment deposition and semi-qualitative estimates of deposition (using the SAS techniques) – when averaged over longer rather that shorter time periods.

Distance decay relationships suggest that sediment related impacts on corals occur relatively near to dredging activities. While changes to turbidity can occur at relatively large distances from dredging activity (>20 km), sediment related coral health exposure metrics dissipated rapidly, with effects limited to less than 3-4 km (Project 4.9.2) and distances of exceedance of the 14 d running mean turbidity thresholds indicative of probable mortality effects on coral (permissive thresholds), in the order of only 500 m from dredging (see Figure 1, Project 4.9.4).

Light attenuation and coral health

Branching corals appear to be quite sensitive to light limitation during dredging, and exposure to low light causes bleaching and partial mortality. While impacts on benthic light levels can be influenced at distances very far from dredging activities (up to >20, or even >35 km from dredging), reduction of light to below levels that can cause sub-lethal stresses on corals occur much closer, with laboratory based thresholds for coral discoloration (EC_{10}, (Bessell-Browne et al. 2017b)) exceeded at Barrow Island only up to ~3.5 km. Light levels low enough to cause elevated mortality of corals occur at distances of only 0.2-1.9 km from dredging activity (Projects 4.9.2 and 4.9.4).

Management Implications

Pre-development surveys

Coral communities can be naturally exposed to a range of disturbance regimes that can correlate with dredging related pressures, resulting in communities that are potentially more, or less susceptible to dredging impacts. Extensive surveys of the receiving benthic communities prior to impact prediction are essential for applying appropriate tolerance thresholds during impact prediction. In particular, care needs to be taken to assess the

relative contribution of massive versus branching corals, as these differ in their primary cause-effect pathways by which dredging impacts will manifest. Given that some dredging related pressures (namely reduced light availability) co-vary with depth, care should be taken to capture the existing coral community across the full range of depths of likely impacted reef areas. Repeated sampling through time and replicate sampling across space of the abundance and/or occurrence of key target taxa is essential to properly quantify natural spatio-temporal variability where a before/after control/impact (BACI) design will be used to assess dredging related impacts. Wherever possible control locations should be identified which are as similar as possible to impact locations both in terms of the type of coral community, the depth of the community, as well as background environmental conditions, as this will greatly improve the ability to definitively partition dredging impacts from natural disturbances.

Where the intention is to use thresholds derived relative to baseline conditions, long time series (>2 years) of water quality data (sedimentation, NTU and/or light) should be collected to ensure the full natural exposure regime is adequately captured. Concurrent assessment of TSS-NTU relationships and light attenuation coefficients should also be collected to provide the necessary data for impact prediction. Data affected by cyclones in the baseline phase needs to be examined on a case-by-case basis if baseline data is used for deriving thresholds. The consideration needs to be whether it is reasonable to assume that cyclones have had extreme effects on the baseline, based on the data collected, but also on the size and proximity of the cyclone.

Impact prediction

Defining zones of impact

Using a combination of the *ex situ* and *in situ* studies, from field observations and bioindicator analyses, analyses of cause-effect pathways and pressure-response relationship and physiological and ecological endpoints, a weight of evidence (WoE) approach was used to define the putative management zones: zone of high Impact (ZoHI), zone of moderate impact (ZoMI) and zone of influence (ZoI, EPA (2016), Figure 1). We suggest that the ZoHI (orange shaded area, Figure 1) be determined as those areas that encompasses distances of exceedance of experimentally derived light based thresholds for increases in the probability of mortality of laboratory based corals (EC_{10}, *(Bessell-Browne et al. 2017b)*, Figure 1 - e), previously calculated thresholds corresponding to a high probability of observing non-zero coral mortality (see *Permissive* thresholds in *Project 4.9.3*, Figure 1 – b, c, & d) and estimated 'no effect distances' for massive *Porites* and branching *Acropora*/Pocilloporidae ((Fisher et al. 2018, Project 4.9.3), Figure 17 – f & g).

The ZoHI is followed by a ZoMI which the EPA defines as the area within which predicted impacts on benthic organisms (corals) are recoverable within a period of five years following completion of dredging activity. We suggest that the ZoMI be determined as the area that encompasses the region immediately outside the ZoHI up to distances where sub-lethal impact thresholds (EC_{10} coral bleaching, Figure 17 - m; ED_{10} for accumulated sediment and mucus occurrence on *Porites*, Figure 17 – j & k), or thresholds indicative of only *possible* coral mortality (*strict* thresholds, (Fisher et al. 2018, Project 4.9.3*)*, Figure 17 – h, i & n) are no longer exceeded (green shaded area, Figure 17).

Beyond the ZoMI there is the ZoI, where changes to water quality may occur, but not to an extent that constitutes a hazard to the underlying coral communities (blue shaded area, Figure 17).

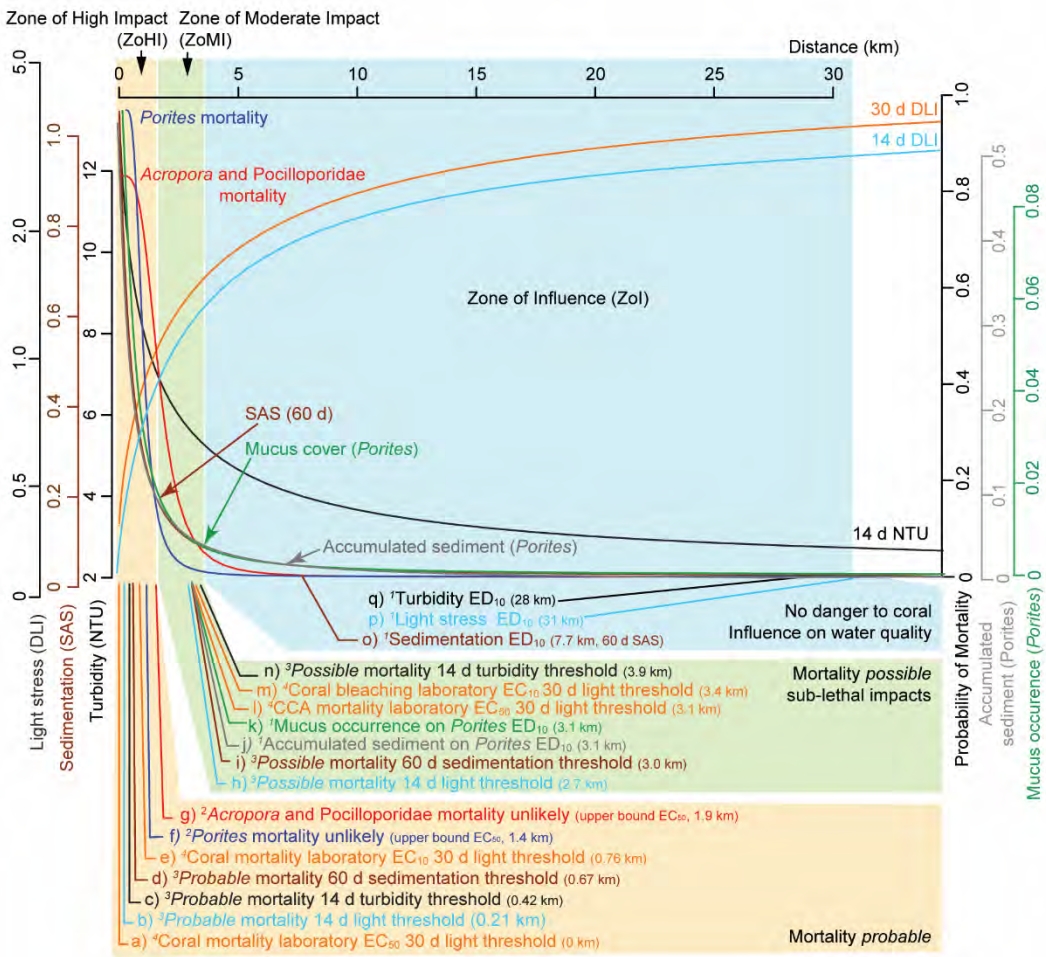

Figure 1. Aggregate distance decay plot for a range of sedimentation and light related water quality and coral health metrics for the first 203 days of dredging during the Barrow Island project. All response variables examined were rescaled to between 0 and 1 and fitted using logistic regression with $\log_{10}$ distance from dredging activity as the. Three zones of impact are defined based on the multiple lines of evidence provided, including: a Zone of High Impact (ZoHI) that encompasses all estimated distances likely to result in elevated coral mortality; a Zone of Moderate Impact (ZoMI) that encompasses all estimated distances likely to lead to sub-lethal impacts on coral communities and *possible* non-zero coral mortality; and a Zone of Influence (ZoI) in which only elevations to water quality metrics occur with no danger to coral. Coloured lines at the base of the plot indicate estimated distances of effect from one of four sources: 1) estimated distances of 10 % effect (ED_{10}, from the maximum to the minimum predicted value at the furthest distance based on the fitted logistic curve, akin to an EC_{10} or LD_{10} in ecotoxicology); 2) estimate upper bound distances where *Porites* massive or *Acropora*/Pocilloporidae branching mortality is unlikely (NEC = ED_{50}); 3) distances of exceedance of field based threshold values (from Fisher et al. 2018); and 4) distances of exceedance of laboratory based threshold values (from Bessell-Browne 2017d).

Impact prediction thresholds

The most robust approach for managing the impacts of future dredging projects is to ensure that **water quality conditions remain within safe bounds across the full hazard profile in the context of both cumulative probabilities of exposure as well as running mean values** spanning different multiple temporal scales. While direct empirical data are available for hazard profiles associated with light reduction (shown in Figure 2), hazard profiles for sedimentation cannot be used because of the issues associated with measuring ecologically relevant sediment deposition levels in the field (Whinney et al. 2017a). In the interim SSC (and/or NTU, shown in Figure 2) can be used as a means of defining hazard profiles that delineate safe and un-safe conditions (to coral) as over the long term there is a good relationship between the two. By fitting non-linear quantile regressions to the lower bounds of all sites <1.9 km ('red-black' sites) and upper bounds of all sites >1.9 km ('blue-green' sites) from the dredging it is possible to obtain corresponding **strict (possible effects)** and **permissive (probable effects)**

thresholds across the full hazard profile for both the cumulative probability distribution, as well as across all durations of running means (Table 1).

Cumulative probability fitted lines are given by:

$$NTU_S = 10^{0.45 \times e^{-0.222 \times log\left(\frac{CP}{1-CP}\right)}} - 1$$

$$NTU_P = 16.1 \times e^{-0.333 \times \log(\text{Duration})}$$

$$DLI_S = \frac{e^{-1.84 \times e^{-0.199 \times log\left(\frac{CP}{1-CP}\right)}}}{1 + e^{-1.84 \times e^{-0.199 \times log\left(\frac{CP}{1-CP}\right)}}} \times 30$$

$$DLI_P = \frac{e^{-2.01 \times e^{-0.228 \times log\left(\frac{CP}{1-CP}\right)}}}{1 + e^{-2.01 \times e^{-0.228 \times log\left(\frac{CP}{1-CP}\right)}}} \times 30$$

Running mean fitted lines are given by:

$$NTU_S = 16.1 \times e^{-0.333 \times \log(\text{Duration})}$$

$$NTU_P = 32.4 \times e^{-0.514 \times \log(\text{Duration})}$$

$$DLI_S = \frac{0.235 \times Duration}{1 + (0.0414 \times Duration)}$$

$$DLI_P = \frac{0.097 \times Duration}{1 + (0.0104 \times Duration)}$$

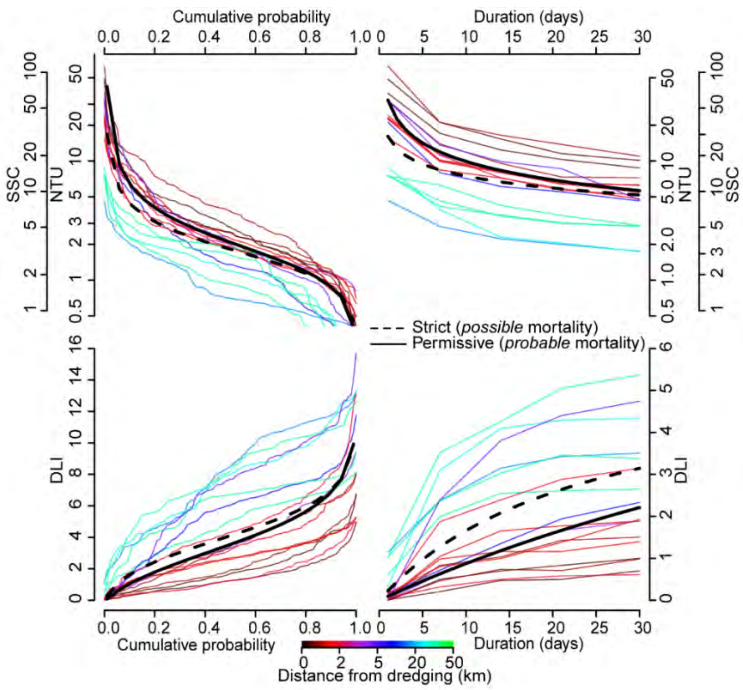

Figure 2. Exposure profiles for cumulative probability (left hand plots) and worst case running means (1 to 30 days duration, right hand plots) for SSCs (mg L^{-1} based on SSC~1.8 × NTU, upper plots) and DLI (mol quanta m^{-2} d^{-1}, lower plots) for the first 203 days of dredging at Barrow Island. Coloured lines show individual profiles across 17 sites of varying distances to the dredging activity, with distance indicated by the colour bar. The interface between acceptable (blue and green lines, will not cause mortality of corals), versus harmful (black and red, may cause coral mortality) hazard profiles can be used to infer the conditions (across the entire hazard profiles) known to be harmful to corals. Fitted equations used for threshold derivation are shown in the right hand side for both Strict (*possible* mortality, Superscript '*S*') and Permissive (*probable* mortality, Superscript '*P*') thresholds.

At the EIA stage, during impact prediction phase, both cumulative and running mean strict (dashed lines) and permissive (continuous lines) can be used to interrogate the coupled sediment transport and hydrodynamic models that predict sediment transport and fate, to estimate whether effects on corals are possible and/or probable respectively. If the predicted (modelled) hazard profiles cross the lines at any stage this would be used to categorize a site (ZoHI and ZoMI or ZoI). If the relationship between the sediment types and particle size and light attenuation properties are known (see Theme 3), then the DLI data can also be used in combination with the SSC data for impact prediction and site classification according to the zoning scheme.

Table 1: Derived *possible* and *probable* coral mortality thresholds across the complete hazard profiles for cumulative probability (A) and running mean durations (B) for managing dredging activities near coral reefs. *P*ossible effects (aka strict)

thresholds are interpolated from the lower bounds of all sites <1.9 km (unsafe, 'red-black' sites, upper distance for potentially harmful effects, orange area Figure 1) and *probable* effects (aka permissive) thresholds are interpolated from the upper bounds of all sites >1.9 km (safe, 'blue-green' sites) from the dredging (see smooth dark black lines, Figure 1). Interpolations at the upper and lower bounds were achieved via non-linear 0.9 and 0.1 quantile regressions. SSC units are mg L^{-1} and DLI units are mol quanta m^2 d^{-1}.

Threshold type		**Possible effects *(strict)***			**Probable effects *(permissive)***		
		>NTU	**>SSC**	**<DLI**	**>NTU**	>SSC	<DLI
Cumulative probability (% days)	90%	0.7	1.3	7.4	0.7	1.3	6.6
	80%	1.0	1.9	6.3	1.2	2.1	5.4
	70%	1.3	2.3	5.5	1.6	2.8	4.6
	60%	1.5	2.8	4.9	1.9	3.4	4.0
	50%	1.8	3.2	4.4	2.3	4.1	3.4
	40%	2.0	3.7	3.8	2.6	4.8	2.8
	30%	2.3	4.2	3.2	3.1	5.6	2.3
	20%	2.8	5.0	2.6	3.8	6.9	1.7
	10%	3.5	6.3	1.7	5.1	9.1	1.0
Running mean (days)	1 d	15.5	27.9	0.4	32.4	58.3	0.1
	3 d	10.8	19.4	1.1	19.9	35.7	0.3
	7 d	8.2	14.7	1.8	13.6	24.5	0.6
	10 d	7.3	13.1	2.2	11.6	20.9	0.9
	14 d	6.5	11.7	2.5	10.0	18.0	1.1
	17 d	6.1	11.0	2.7	9.2	16.5	1.3
	21 d	5.7	10.2	2.9	8.3	15.0	1.5
	28 d	5.2	9.3	3.1	7.3	13.2	1.8
	30 d	5.1	9.1	3.1	7.1	12.8	1.9

Relating impact predictions to threshold pressure metrics and in situ monitoring

All water quality metrics examined were relatively strong predictors of the probability of non-zero coral mortality, thus regardless of the metric used water quality conditions should be considered robust predictors of dredging impacts. However, care must be taken to ensure that any thresholds used for predicting impacts are accurately represented by dredge plume modelling outputs. Some general recommendations include:

- Estimated running mean values can vary with the statistical treatment of the data (such as geometric versus arithmetic means). How data are handled, in the context of transformations and averaging, must be carefully considered and always clearly reported, both in the context of threshold derivation, as well as handling of hydrodynamic modelling outputs;
- TSS-NTU relationships are highly variable in space and time, especially where sediment regimes are being modified through dredging activities. Adequate data must be collected to ensure dredge plume modelling can accurately predict NTU values where these are used as thresholds, and are intended for use in dredge management;
- Dredge plume models must accurately predict benthic light across the study domain, given the importance of this variable in driving some aspects of coral reef communities.
- Analyses indicated that for turbidity and light two week running means were appropriate time scales for assessing impact, but for deposition longer running mean time scales (60 days) were better predictors. However, given the interaction between threshold intensities and durations, and the difficulty in distinguishing between the relative importance of acute and chronic effects, predictions should be examined across the full suite of hazard profiles generated (see above, Figure 2).

Management triggers

Triggers for different management actions will depend implicitly on the spatial zone in which the trigger is exceeded, as they should relate in a direct way to the thresholds that are used for delineating zones of impact. Thus the distinction between thresholds for probable (p*ermissive)* and possible (*strict*) mortality of corals (Table 1 above) will directly apply as management triggers within the appropriate spatial zone, but the management actions that each initiate are zone dependent (Table 5).

Table 2: Relationship between thresholds for possible and probable mortality (see Figure 2 & Table 1) and indicative management actions required for their respective exceedance across different management zones.

	Zone of High Impact (ZoHI)	**Zone of Moderate Impact (ZoMI)**	**Zone of Influence (ZoI)**	**Outside Zone of Influence**
Possible mortality (*strict* threshold)	No action	Monitor operations	Ensure improved water quality	Stop dredging
Probable mortality (*permissive* threshold)	Monitor operations	Ensure improved water quality	Stop dredging	Stop dredging

Within the ZoHI total coral loss is generally allowed and there is no reason to act following exceedance of *possible* mortality thresholds. Exceedance of *probable* mortality thresholds may trigger active monitoring of operations. Within the ZoMI exceedance of *possible* mortality should only result in the monitoring of operations, whereas exceedance of *probable* mortality thresholds should initiate actions to ensure improved water quality conditions. Within the ZoMI exceedance of *probable* mortality thresholds would not require halting dredging, assuming that some mortality in the ZoMI is allowable under the project conditions. Within the ZoI exceedance of *possible* mortality thresholds indicates management actions should be initiated to improve water quality conditions, and exceeding *probable* mortality thresholds could result in a stop dredge situation (provided the exceedances are not entirely due to natural turbidity events unrelated to dredging) and initiate benthic monitoring to assess if there has been non-compliance within the ZoI.

During dredging programs 1–28 d running mean SSCs or DLI values can be plotted together with the guideline values giving dredging proponents information on whether effects are possible or probable at both short term (days) and longer term (weeks) intervals. For example, we show nephelometrically derived SSCs (mg L^{-1}) collected at a water quality monitoring station ~200 m from dredging during the 530 d Barrow Island project (Figure 3), with 1–28 d running mean values over two × 10 d periods during the dredging where there were clearly defined short duration increases (spikes) in turbidity (Figure 3B and C). In the first period the mean daily SSC increased from 10 mg L^{-1} to 20–40 mg L^{-1} on days 3–6, close to or exceeding the 'possible thresholds' value, but over the remaining 4 days decreased to <5 mg L^{-1}. As a result, the 14 d running mean value was less than the possible effect guideline (11.7 mg L^{-1}, see Table 1, Figure 3B). In the second and much more substantial increase in turbidity, the mean daily SSC increased exceeded 50 mg L^{-1} on days 1–4 exceeding the 'possible' and 'probable' thresholds' values (Figure 3C). From day 5 onwards the SSCs decreased but were still >10 mg L^{-1} for the remainder of the period. As a result of the short term (acute) period of exceptionally high SSCs (>100 mg L^{-1}) and the sustained 10 d period of SSCs >10 mg L^{-1}, the 14 d running mean value exceed the possible (11.7 mg L^{-1}) and possible (18 mg L^{-1}) thresholds considerably (see Table 1, Figure 3C). It is important to note for contextual purposes that the site was <200 m from the dredging (within the area of predicted and approved impact on corals) and the analysis in Figure 3A shows it included the highest 1 day mean SSC in the 530 d dredging program. Similar peaks were observed at other sites and this spike is likely to have been caused by a storm combined with resuspension of loose unconsolidated sediments.

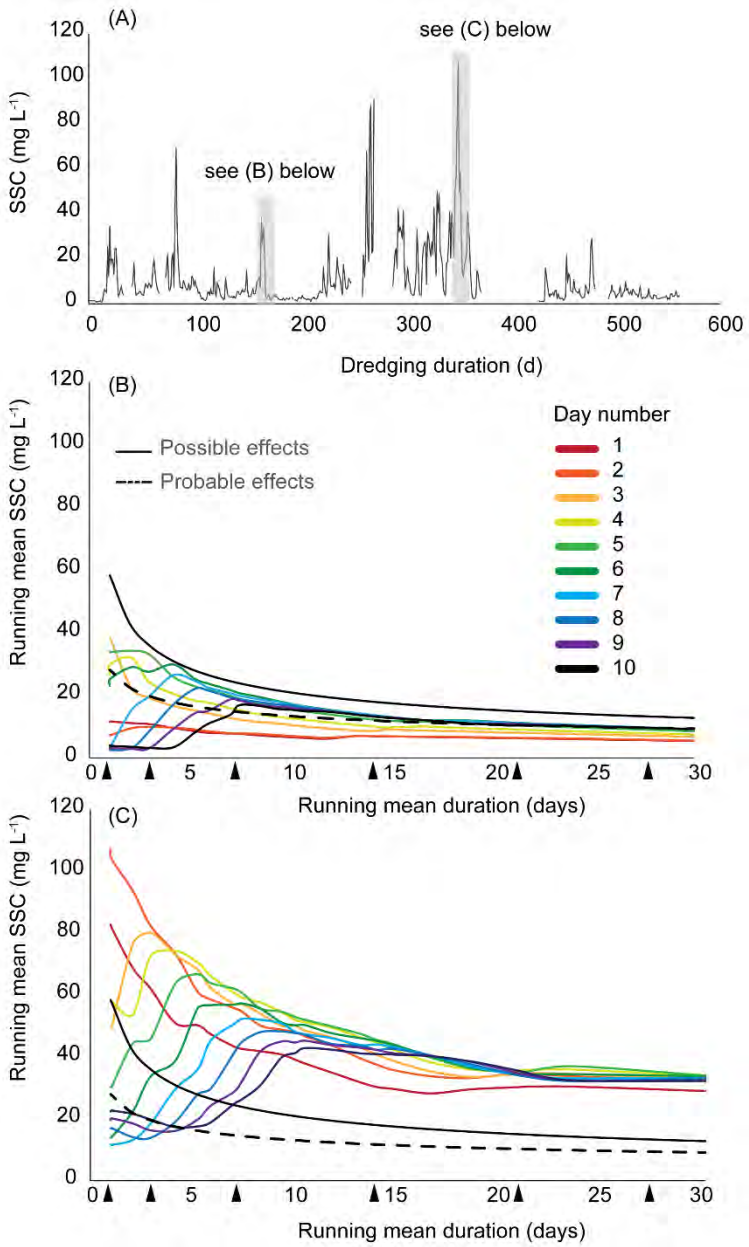

Figure 3. (A) Nephelometrically derived SSCs (mg L^{-1}) at a water quality monitoring site ~200 m from dredging during the Barrow Island dredging project. (B) and (C) show the 1–28 d running mean values calculated for each of 10 days spanning two turbidity events, one starting on day 166 (1 Nov 2010, B) and the other on day 349 (3 May 2011, C). The solid and dashed black lines show the derived *Possible* and *Probabl*e effect thresholds across the 1-28 d hazard profile (See Fig 2 and Table 1).

One of the primary considerations associated with development of the guidelines and the running means approach, as discussed in (Fisher & Jones 2018), is that the total pressure must be accounted for, i.e.:

- Dredging-related pressure is calculated for light and SSCs in absolute terms i.e. a running mean of $\boldsymbol{x}$ mg L^{-1} (or mol quanta m^2 d^{-1}) over a period $\mathbf{y}$ days, and not as days exceeding a certain value i.e. number of days >$\boldsymbol{x}$ mg L^{-1} (or <x mol quanta m^2 d^{-1}) – which would not allow determination of total load as how far the threshold was exceeded by is not known;
- Pressure is calculated as a running mean in absolute terms (i.e. $\boldsymbol{x}$ mg L^{-1}) and not relative to reference sites (i.e. $\boldsymbol{x}$ mg L^{-1} over a reference site value of $\mathbf{y}$ mg L^{-1}). From a physiological perspective corals do not discriminate between dredging-related and natural resuspension events and basing thresholds in comparison to reference sites would also not allow the total accumulated pressure to be monitored over the time frame considered.
- Calculating dredging-related pressure in absolute terms over clearly specified running mean time intervals allows more precise calculation of the accumulated load than coarser time averaging approaches – for example >$\boldsymbol{x}$ mg L^{-1} >20% of the time (where multiple consecutive exceedances may have much greater

consequences than multiple exceedances interspersed with periods of otherwise good water quality).

Impacts of dredging on water quality are highly ephemeral and spatially complex, with changes to water quality occurring across both acute scales (changes to the extremes) and chronic scales (changes to long term conditions, (Fisher et al. 2015, Jones et al. 2015)). Time is an important consideration in threshold development and running means analysis (coupled with the *possible* and *probable* mortality guidelines) have temporal components explicitly integrated into the analysis at multiple (rather than single) time periods accounting for both acute and more chronic effects.

Maintenance dredging projects in Western Australia are typically 4–10 weeks, slightly longer than in Queensland where the average maintenance dredging duration is generally 2–4 weeks (Ports Australia 2014). The duration of capital dredging projects in Australia has ranged from a few weeks to up to 2.5 years, although most extend for 4–6 months; however, even within extended capital dredging projects, dredging often occurs across multiple locations (for example along a channel) and dredging pressures may effect individual areas for a much shorter time. Thus basing thresholds on a 1–28 d time frame is likely to be functionally useful for both shorter (maintenance) and longer (capital) dredging campaigns.

We suggest that across the 1–28 d running mean period additional significance should be given to 7, 14 and 28 days and special emphasis placed on the 14 time period, for the following reasons:

- (Fisher et al. 2017), calculated running means across 1–60 day time scales for the Barrow Island data set and found that a 14-day running mean showed the strongest relationships with the probability of non-zero coral mortality for both turbidity and light stress;
- The time-frames allow dredging proponents enough time to plan and enact adaptive management activities if required (such as moving dredges, changing dredging modes or production rates etc);
- Having a comparatively short time component associated with a threshold i.e. a running mean over weeks as opposed to months, means that if all dredging activities were reduced or to cease entirely, the running mean values would respond (i.e. improve) relatively quickly (as can be seen in Figure 3B);
- Natural events such as storms can be reliably predicted out to around 7 d by the Australian Bureau of Meteorology Next Generation Forecast and Warning System, and natural turbidity generating events such as storms or weather fronts are thus more foreseeable;
- Similarly, if there are cyclical periods associated with turbidity and light availability (such as with the 14 d spring-neap cycles), proponents can factor natural hydrodynamics into forward planning and decision-making purposes.

From an operational perspective, depending on the zone, and also the risk aversion of the proponent, crossing the possible or probable profiles (at any point), could result in management-considerations (such as changing dredging modes or production rates etc, moving dredges, to stopping all dredging). For example, exceeding a DLI or SSC *possible* effect running-mean threshold over any time period in sites designated as within a ZoI should probably prompt a management action. We suggest a management aim would be to be compliant to the zone at the 7, 14 and 28 day time frames even though the thresholds may be crossed intermittently between these times.

Other management considerations

Proponents should implement measures to ensure that triggers are not reached. This can be achieved using control charts or similar strategies that report on water quality conditions over shorter time scales than the actual trigger and provide an early warning of impending exceedance. For example, if management triggers are based on 14 day running mean turbidity, exceedance of the same trigger over a 7 day running mean would indicate that water quality conditions may be nearing the trigger value, providing an early indicator that management actions may be required.

Setting thresholds as a multiplier of baseline conditions (e.g. mean × 2.3 (Strict, Tier 1) or × 3 (Permissive, Tier 2) NTU) to allow for differences in background environmental conditions should be adopted in preference to the

use of highly unstable extreme percentile values, as the coral-mortality derived thresholds at barrow Island were nearly at the 100th percentile for most sites, and maximum values will be highly sensitive to subtle differences in the conditions observed during baseline. Setting thresholds based on either approach is not recommended because corals growing in marginal environments (i.e. turbid reef zones) may be closer to their physiological limits. For corals growing under extremely clear conditions, adjustments based on either percentiles or multipliers may prove overly conservative. Efforts should be made to verify tolerance limits of corals growing in very different environments. Where this is not feasible, conservative thresholds should be set and adaptive management strategies should be adopted that include *predictive links* monitoring that will aid in refining thresholds throughout the proposed campaign.

Deriving thresholds that avoid false exceedances poses a significant challenge. Collecting long run time series during the baseline phase at all sites of interest are essential to ensure that sites are adequately categorised into the appropriate background water quality regime, as well as to properly capture the full range of temporal variability and exposure of the natural water quality regime of each site. Water quality data (turbidity, PAR and sediment deposition) that is affected by cyclones during the dredging periods should be included in the assessment of water quality guidelines, as it is the combined water quality conditions that will impact corals, regardless of the actual cause of the poor water quality conditions. Similarly, absolute value (i.e. x mg L^{-1} for y days or over period y) should be used for guidelines as opposed to relative (to reference locations) values (i.e. x mg L^{-1} for y days or over period y, over a background of z mg L^{-1}). Regardless of how thresholds are derived, false exceedances of water quality thresholds are inevitable if thresholds are to have any statistical power to detect potential impacts. A multiple lines of evidence approach should be adopted, where, in the case of an exceedance, ancillary data and information are examined at the time of the exceedance and this information is factored in when management actions are being evaluated. If it is clear that the exceedance is due entirely to natural events and is not being exacerbated by the dredging activities, modifying dredging activities may be unnecessary, even in the event of an exceedance.

Tolerance levels in the face of cumulative impacts (bleaching and/or long term dredging campaigns – see below) are probably lower, and management triggers should be adjusted accordingly where cumulative impacts occur and/or the dredging campaign is of extended duration. Our analyses found that depending on the severity of their impact on benthic light and sedimentation, suspended sediments can have both negative and positive effects on corals during periods of thermal stress. Low to moderate reductions in light alleviate coral bleaching through shading, meaning lower overall mortality of coral (antagonistic cumulative impacts). Conversely, at high sediment loads any positive effect of reduced bleaching is far outweighed by higher levels of mortality associated with sediment related stress, and actually results in higher overall levels of mortality of coral (synergistic cumulative impacts). The findings suggest that while there may be no reason to limit sediment generating activities (such as dredging) near coral reefs if the water quality conditions can be maintained above the transition from antagonistic to synergistic impacts (~1.2 14 d running mean DLI), at higher sediment loads the cumulative impacts of suspended sediments and thermal stress may be much worse when they occur concurrently.

Monitoring

Water quality thresholds represent a relatively robust means of guarding against coral mortality during dredging, and their use as prior warning management triggers should be encouraged. During monitoring and reporting care must be taken to ensure that metrics used accurately represent the thresholds adopted during dredge plume modelling that were used to predict impacts (i.e. management triggers and their associated monitoring must reflect the impact assessment thresholds in a clear and accurate way). Noting that many statistical summaries of time series data, including running means, can vary with the statistical treatment of the data (such as geometric versus arithmetic means, see above). How data are handled, in the context of transformations and averaging, must be carefully considered and always clearly reported. Wherever possible, monitoring programs based around NTU and/or benthic PAR should periodically collect the necessary additional data to develop TSS-NTU relationships and light attenuation coefficients, given their importance in linking predictive dredge plume

modelling to impacts on benthic communities.

In addition to monitoring of water quality trigger metrics, monitoring of coral community health should also be undertaken at periodic intervals throughout dredging at both impact and reference locations. Time series data on benthic community percentage coverage is essential for assessing temporal variability and is key to ensuring that any drivers resulting in changes to the community can be attributed appropriately. Collection of coral health data from within zones of predicted high impact is invaluable in deriving *in situ* thresholds, and should be encouraged for all future dredging projects as this enables validation of laboratory experiments and furthers our understanding of how thresholds may vary across different settings. Making all such monitoring data available for scientific study will maximise the return on investment by proponents, as well as result in positive outcomes that offset the inevitable environmental harm.

Sub-lethal impacts such as bleaching, colour score index and mucus prevalence may be expected up to distances of approximately 2-times the distances of realised impacts. Monitoring sub lethal indicators may provide extra insurance against lack of compliance, by delineating the spatial bounds of anticipated realised impacts.

Key residual knowledge gaps

The thresholds derived from the Gorgon dataset are likely specific to the mortality and exposure profiles that occurred at Barrow Island and how this relates to the natural sediment exposure regime. Further studies exploring variation in the sensitivity of individual taxa and communities to dredging related pressures would provide insight into the extent to which the derived thresholds may apply more generally, or if location specific thresholds are required. Specifically, more research is required to properly understand how management thresholds should be appropriately adjusted for coral communities across broad gradients in natural and anthropogenically modified background exposure regimes.

The knowledge gained through the “predictive links” monitoring undertaken in addition to the required compliance monitoring at Barrow Island is substantial. Science is an iterative process and similar efforts, which represent only small additional costs once extensive compliance monitoring is underway, should be encouraged wherever practicable for future dredging campaigns. Each new dredging project provides an opportunity to test the generality of thresholds and explore further complexities in the appropriate management of dredging operations in the vicinity of coral reef habitat.

Corals are a highly diverse group that differ substantially in terms of their life history traits, including polyp size, reproductive characteristics and morphology. They are known to also vary in terms of their susceptibility to thermal stress, and our experimental and *in situ* data also suggest they vary substantially in response to dredging related stressors. Models attempting to understand the temporal and spatial scales of potential impacts of dredging on nearby coral communities require clear data on the relative sensitivity of the constituent species. To make modelling of communities tractable, it is essential that functional trait groups are delineated. These groups must capture not only the key biological traits important in the context of population modelling, but also sediment and light stress sensitivities. Such traits could include the surface rugosity of colonies, polyp size, ability to switch between autotrophy and heterotrophy, along with the amount of energy reserves. Functional trait groups that capture coral sensitivities to dredging related pressures will be useful in helping guide management recommendations for different coral communities, as well as simplify modelling highly speciose, complex coral communities.

Our analyses clearly indicated that deposition based metrics are strong predictors of coral mortality. While the sensory logger based SSSD thresholds developed here provide information on sediment deposition over appropriate time frames (days), recorded values are dependent on the physical properties of the sensor surface. There is a pressing need to develop instrumentation to measure sediment deposition rates with appropriate sensitivity and adequate temporal resolution, and to further derive relevant tolerance thresholds for the resulting metrics of deposition.

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Projects

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Synergistic and antagonistic impacts of suspended sediments and thermal stress on corals

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Understanding pressure pathways and their cumulative impacts is critical for developing effective environmental policy. For coral reefs, wide spread bleaching resulting from global warming is occurring concurrently with local pressures, such as increases in suspended sediments through coastal development. Here we examine the relative importance of suspended sediment pressure pathways for dredging impacts on corals and evidence for synergistic or antagonistic cumulative effects between suspended sediments and thermal stress. We show that low to moderate reductions in available light associated with dredging may lead to weak antagonistic (less than expected independently) cumulative effects. However, when sediment loads are high any reductions in mortality associated with reduced bleaching are outweighed by increased mortality associated with severe low light periods and high levels of sediment deposition and impacts become synergistic (greater than what would occur independently). The findings suggest efforts to assess global cumulative impacts need to consider how pressures interact to impact ecosystems, and that the cumulative outcome may vary across the range of realised pressure fields.

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The world's coral reefs are under increasing pressure from a range of global and local threats[1,2], posing challenges for environmental management and regulation of these valuable ecosystems[3–5]. With nearly the entire ocean affected by multiple stressors simultaneously[2], an understanding of how they interact is critical to predicting their cumulative impacts. While most assessments of cumulative impact assume additive effects[6], stressors may in fact interact such that their combined impacts are greater (synergistic) or even less (antagonistic) than might be expected in isolation[7].

A key global pressure is climate change-induced increases in sea surface temperature, leading to recurrent mass coral bleaching[8,9], potentially transforming coral reef assemblages[10,11]. In addition, elevated suspended sediment concentrations (SSCs), through runoff[12,13], resuspension[14] and dredging activities[15] are an important local source of reef degradation in coastal waters. Suspended sediments can impact corals through three interrelated pressure pathways[16], including: interfering directly with heterotrophic feeding[17,18], attenuating light and reducing rates of algal symbiont photosynthesis[19], and increased sediment deposition leading to smothering[20,21].

Using data from a large-scale dredging project undertaken on the reefs around Barrow Island in north Western Australia, we examine the relative importance of the three suspended sediment pressure pathways, along with thermal stress and associated coral bleaching, in predicting coral mortality. The 530 day Barrow Island dredging project coincided with a mass coral bleaching event[22] associated with a week of unusually warm water temperatures in the region[23]. The dredging project included extensive water quality and coral health monitoring, before, during and after completion of the dredging activities, at locations covering a gradient of dredging-related exposure, from within a few hundred meters up to more than 30 km from the dredging activities. This monitoring resulted in one of the largest datasets ever collected on individual tagged coral colonies exposed to dredging activities, and included estimates of dredging-related pressure (suspended sediments, available light, and sediment deposition), thermal stress (water temperature), along with measurements of coral health (~2-weekly measurements of ~1500 individual coral heads) throughout an entire 1.5 year dredging campaign. The fact that the data set captures not only a complete gradient of dredging-related pressure, but also temporally spans a thermal stress event, provides a unique opportunity to look at the cumulative impacts of these two important stressors in situ, and examine evidence for synergistic or antagonistic cumulative effects.

The analyses show that, depending on the severity of their impact on benthic light and sedimentation, suspended sediments may have both negative and positive effects on corals during periods of thermal stress. Low-to-moderate reductions in available light from suspended sediments can reduce the incidence of coral bleaching, and may reduce overall coral mortality, particularly for branching corals. However, when sediment loads are high any reductions in bleaching incidence are outweighed by increased mortality associated with severe low light periods and high levels of sediment deposition. The outcome is that under low sediment loads the cumulative impact of suspended sediments and thermal stress may be less than expected (antagonistic), whereas at high sediment loads the overall impact is greater than when these stressors occur in isolation (synergistic). The results highlight that while pressures such as thermal stress associated with climate change can only be managed at a global scale, management of local pressures may, in some cases, have the capacity to modify their overall impact.

Results

Incidents of coral bleaching and mortality at Barrow Island. Incidents of both whole and partial mortality of branching (*Acropora* spp. and Pocilloporidae) and massive (Poritidae) corals was observed as a result of both the combined impacts of dredging activity, and thermal bleaching (Fig. 1). For the branching corals whole coral mortality was 27% (65 of 241 of colonies dying) at the end of the project (after the combined impacts of both dredging and the bleaching event), with whole colony mortality distributed across sites both near and far from dredging (Fig. 1a). However, 26% of colonies (63 of 241) showing no mortality at all (Fig. 1a). For massive corals whole colony mortality was < 1% (1 of 400 of colonies dying) by the end of the project, but only 11% (45 of 400) of colonies showed no mortality at all (Fig. 1b), suggesting that small to moderate levels of partial mortality were common for this growth form. Whole colony mortality usually occurred across more than one fortnightly survey period, with only two individual partial mortality events between consecutive surveys representing complete loss of the colony (partial mortality event ~1, Fig. 1c, black open circles). Site mean partial mortality values were quite variable, ranging from near zero to up to 3 to 4% of the colony surface per fortnight, but were generally highest at sites nearest to dredging (Fig. 1c, d, red line). The highest proportion of colonies showing bleaching (>30%) were associated with warmer temperatures and were generally at sites father from the dredging activities, although moderate and low levels of bleaching (<30%) occurred across sites both near and far from dredging (Fig. 1e, f).

Relative importance of mortality pathways. Both our structured equation and full subsets analyses exploring suspended sediment and thermal pressure metrics and stress indicators showed that the mortality of branching and massive corals was driven by both temperature and bleaching, as well as dredging-related pressures (Fig. 2). This highlights the complex interaction between dredging and thermal stress that occurred during the dredging project. There was little evidence that SSC per se was a strong predictor of mortality, showing a low total standardized effect size (Fig. 2a) and low summed AICc weight values (Fig. 2b, c). Thus, while SSC is directly responsible for generating both low light and high sediment deposition stress, it is not an important pressure independently, providing strong field support for the findings of recent laboratory studies[24]. Loss of benthic light was the strongest predictor of the incidence of partial mortality in branching corals (Fig. 2b) and was also important in predicting the amount of live tissue loss in massive corals (Fig. 2c). Periods of low light and darkness limit photosynthesis, which is considered detrimental as symbiont photosynthesis can provide corals with up to 90% of their daily energy requirements[25,26]. Reduction in phototrophic energy generation potentially increases reliance on energy stores, which would gradually decline over time, resulting in mortality once sufficiently depleted[27]. Sediment deposition was important in predicting partial mortality incidence in massive corals (Fig. 2b), but there was no evidence sediment deposition directly impacted branching corals (Fig. 2b). While coral colonies employ a range of active and passive removal mechanisms to clear sediment[28–30], when deposition rates exceed these removal mechanisms, more rapid necrosis of tissue results[21,31].

Predicting bleaching of branching and massive corals. The probability of bleaching was strongly driven by an interaction between temperature and light for both branching and massive corals (Supplementary Table 2, AICc weight = 1). Bleaching only occurred under conditions of high thermal stress (daily mean

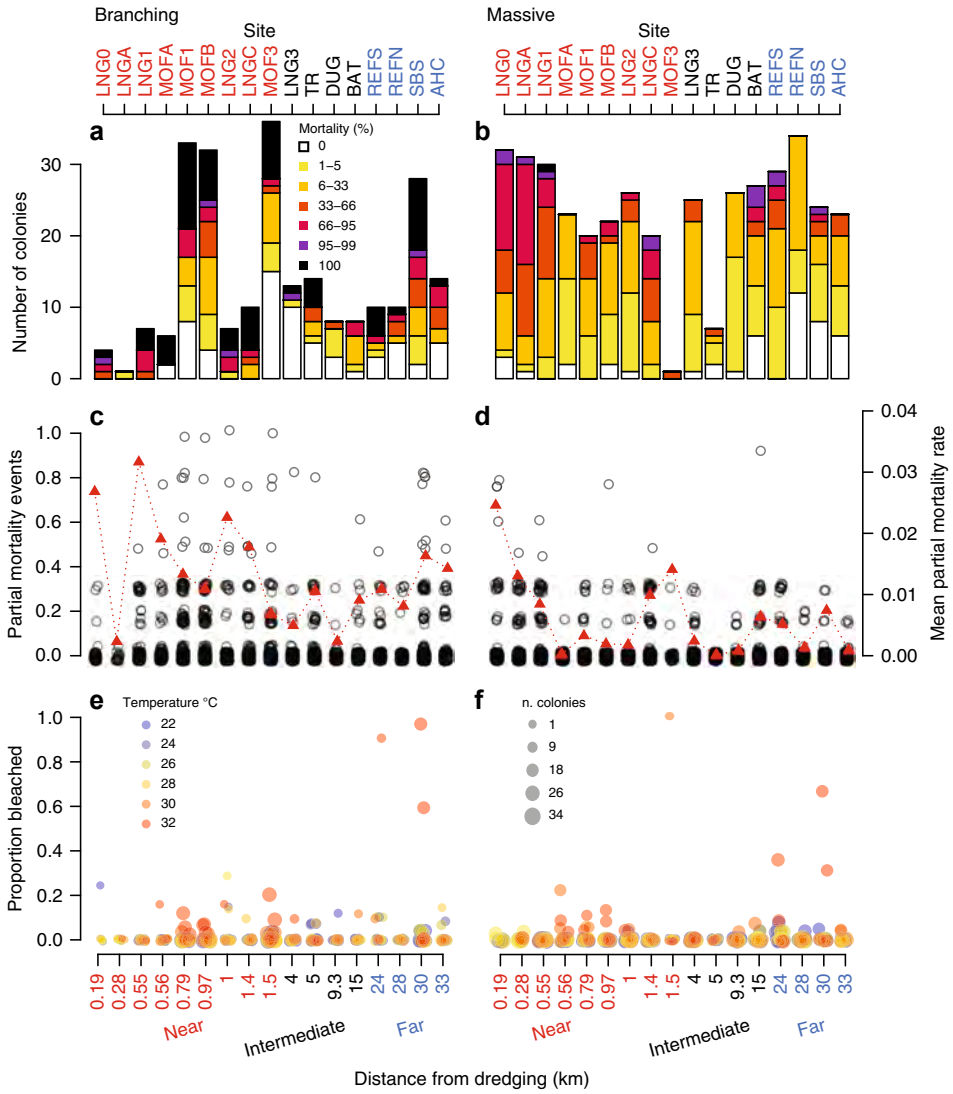

Fig. 1 Mortality and bleaching incidence across 17 sites at Barrow Island. The frequency of branching (left hand panels) and massive (right hand panels) coral colonies exhibiting each of seven mortality scores at the end of the dredging project (**a**, **b**), the magnitude of individual partial mortality events between each survey (~1–3 weeks) (open black circles, **c**, **d**), mean partial mortality (red triangles, **c**, **d**); and the proportion of colonies exhibiting bleaching between each field survey (**e**, **f**), with colour representing the worst case temperature value over the survey period and size indicating the number of observed colonies. X-axis label colours indicate three categories of sites based on previously established patterns in the severity of water quality conditions, including: near (<1.5 km), intermediate (4–15 km) and far (24–32 km) from dredging

temperature > 29 °C) at high light values (> ~4 mol photons m^{-2} per day, Fig. 2a), a phenomenon that has been observed previously[32,33]. It appears that the reduced light levels associated with the suspended sediments generated during dredging may have alleviated the extent of bleaching, a finding consistent with recent laboratory[34] and field[35,36] studies showing lower incidence of bleaching on inshore and/or more turbid reefs and supporting the idea that such reefs may act as refuges under ocean warming[37]. While the interaction between light and temperature was evident for both types of corals, there was a higher overall probability of observing bleaching for branching than for massive corals (Fig. 3a, b), which is consistent with studies examining relative coral sensitivities to thermal stress[38].

Predicting mortality of branching and massive corals. The incidence of any partial mortality for branching corals was best predicted by an interaction between bleaching status, along with both temperature and light (Supplementary Table 2). Once a coral showed signs of thermal bleaching, the probability of observing partial mortality in branching corals was high (around 40%; Fig. 3e). Both light and temperature strongly influenced the incidence of partial mortality events for branching corals, and this is particularly strong for corals showing no prior signs of bleaching (Fig. 3c, Supplementary Figure 4a). At high light levels (> 1.9 mol photons m^{-2} d^{-1}) the incidence of partial mortality increases to values of around 30–40% at the highest temperatures (>31.5 °C), but has no negative impact on corals in the absence of thermal stress (Fig. 3c). Conversely, across the range of observed temperatures low light conditions strongly increase the incidence of partial mortality for unbleached branching corals, with partial mortality events as frequent as 40% under extremely low light conditions (<0.24 mol photons m^{-2} d^{-1}) at low temperature (22–28 °C), and as high as 60% at high temperature (>30 °C, Fig. 3c). Overall, the greatest incidence of mortality of branching corals (both bleached and unbleached) occurred under conditions of both high temperature stress and extremely low light (Fig. 3c, e). The findings support experimental studies suggesting that mortality risk from bleaching is closely associated with energy status in corals[34].

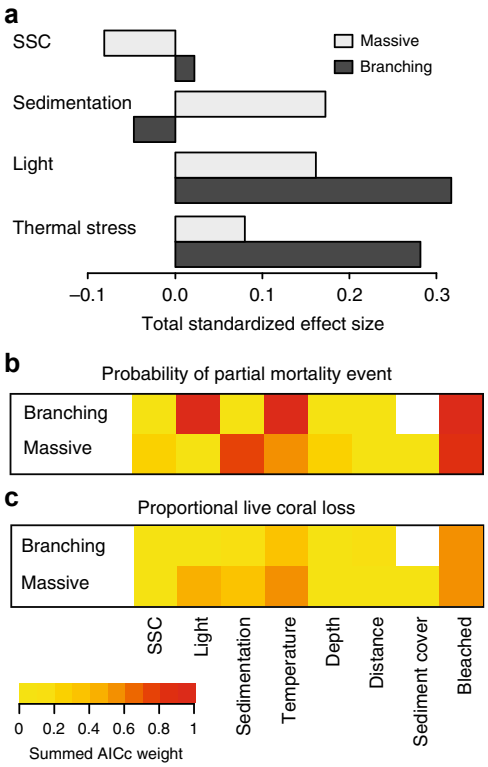

Fig. 2 Relative importance of thermal stress and dredging mortality pathways. Total standardized effect size (**a**) based on summed path coefficients for suspended sediment concentrations (SSC), sedimentation, light, and thermal stress; and summed AICc weight values from a full subsets additive regression analysis for the probability of observing any partial mortality (**b**) and proportional live tissue loss (**c**, Methods 'Predicting coral mortality')

The incidence of partial mortality events for massive corals was higher in bleached corals, but also increased with sedimentation in both bleached and unbleached corals (Fig. 3d, f; Supplementary Figure 4b). There was also evidence that the incidence of any partial mortality also increased with temperature (Fig. 3d, f) and only limited support for any interaction among predictors (Supplementary Table 2). The probability of any partial mortality occurring increased by about 40% from the lowest to highest levels of sedimentation, and was on average about 20% higher for corals showing evidence of thermal bleaching (Fig. 3d, f).

When partial mortality events did occur, the proportion of live tissue lost was only weakly predicted by the pressure metrics examined (Fig. 2c, Supplementary Table 2). Bleaching appeared to have some effect, with the amount of live tissue lost about 30% higher for corals showing evidence of thermal bleaching in both branching and massive corals (Fig. 3g, h). For massive corals the proportion of tissue loss also increased with decreasing light levels (Fig. 3h) and/or sedimentation (Supplementary Figure 4d). Overall, for branching corals the results suggest a high level of mortality associated with thermal bleaching that is further exacerbated by dredging-related stress. Bleached massive corals, on the other hand, only show levels of mortality similar to that of branching corals whilst also under conditions of high dredging-related pressure. Levels of mortality of thermally bleached corals depend on the severity of thermal stress[10]. The lower resilience of bleached massive corals under conditions of high sedimentation and low light likely reflect the reduced capacity for bleached corals to remove sediments[39].

Cumulative impacts of suspended sediments and thermal stress. Using a Monte–Carlo simulation approach based on Bayesian fits of the best models we predicted fortnightly partial mortality under the following three scenarios: (1) thermal pressure; (2) dredging pressure (black lines, Fig. 4a, b); and (3) cumulative thermal and dredging pressure (red lines, Fig. 4a, b). From the thermal pressure alone and dredging pressure alone scenarios we also calculated a theoretical 'additive' prediction (blue lines, Fig. 4a, b; see "Methods" section 'Estimating individual and cumulative impacts'). For branching corals, there are a range of light values where the cumulative impacts on coral appear to be below the theoretical additive level (red line is below the blue line, Fig. 4a) and appear slightly antagonistic (i.e., less than in isolation, Fig. 4c). This weak antagonistic effect reflects the reduced incidence of bleaching associated with the shading effect of low light conditions during dredging. Peak probability of antagonistic cumulative impacts occurs at 3.5 mol photons m^{-2} per day for branching corals, and indicates the optimal light threshold for minimising thermal stress. However, at 1.2 mol photons m^{-2} per day simulated cumulative impacts on coral loss cross the theoretical additive level (red line is above the blue line, Fig. 4a), indicating the impacts are now synergistic (i.e., worse than in isolation, Fig. 4c). This occurs because elevated mortality associated with very low light conditions (and presumably the associated reduced energy status[34]) exceeds any gain associated with reduced incidence of bleaching.

For massive corals, the reduced coral loss through potential shading effects is less than for branching corals, with limited support for antagonistic effects at any light values (Fig. 4b, d). This occurs because high sedimentation and low-light conditions increase the incidence of mortality as well as the amount of coral tissue lost in both bleached and unbleached massive corals, and bleached massive corals do not necessarily have a high probability of mortality when dredging conditions are benign. Both reduced light[34] and increased sedimentation are likely to decrease the energy status of corals, particularly when the suspended sediments themselves infer no nutritional value, as in the case when dredging material of very low organic content, such as occurred at Barrow Island. For massive corals, the transition from antagonistic to synergistic cumulative impacts occurs at 3.1 mol photons m^{-2} d^{-1}. While this intersection point occurs at a relatively high light value for massive corals (this value is naturally exceeded in the absence of dredging during winter months, see Supplementary Figure 2a), the transition to synergistic cumulative impacts is very gradual (Fig. 4d), with an 80% probability of synergistic impact not reached until much lower light levels (i.e., much less than the 1.2 mol photons m^{-2} per d threshold for branching corals, Fig. 4a).

Discussion

At high dredging-related pressure, our models suggest strong evidence for synergistic impacts between sediment-related stress from dredging and thermal stress on corals, with overall mortality highest when these impacts occur together. Depending on the sources of the suspended sediments, expected synergistic impacts may only occur over very limited spatial scales. We estimated partial mortality as a function of distance from the dredging activities using the fortnightly water quality conditions observed at Barrow Island (Fig. 4e, f, Methods 'Distance of impacts'). Estimated distances of 50% effect on mortality under thermal stress occurred at 0.45 km (0.24–2.5 km) for the branching corals (Fig. 4e) and 1.0 km (0.19–8.1 km) for massive corals (Fig. 4f). In the absence of thermal stress distances of 50% effect on mortality were 0.92 km (0.19–4.3 km) for branching corals (Fig. 4e) and 0.29 km (0.19–0.50 km) for massive corals (Fig. 4f). This suggests

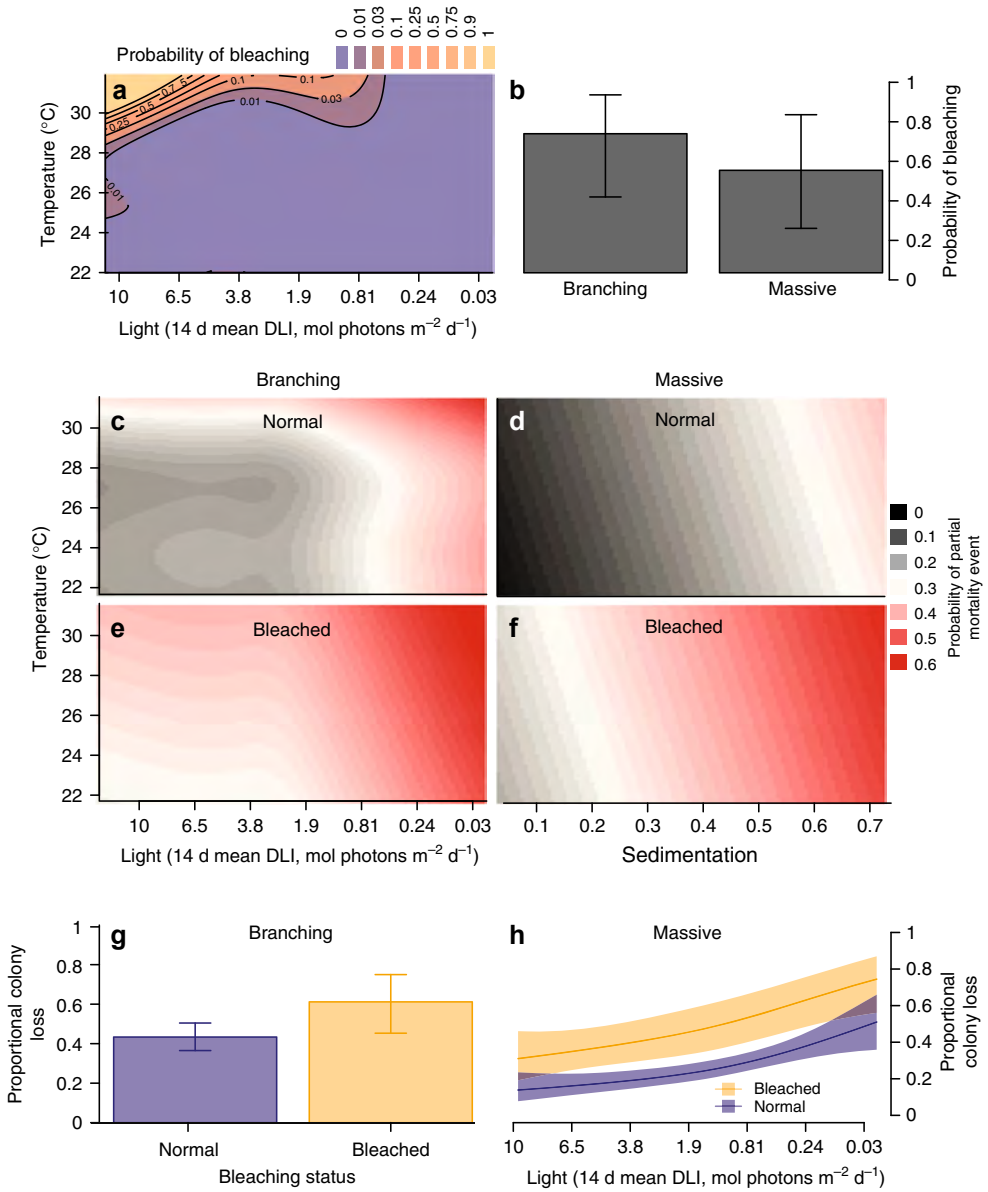

Fig. 3 Relationships between thermal and dredging-related stress metrics and coral health. Mean predictions from the best fit models (lowest AICc, Methods 'Predicting coral mortality') for the probability of bleaching (**a**, **b**), probability of mortality (**c**–**f**), and the proportional loss of coral tissue given mortality (**g**, **h**). Error bars are 2 × estimated se, as reported from the gamm4 model fit

that while cumulative synergistic impacts may increase the distances of expected mortality of massive corals, overall most mortality is occurring within a few hundred meters from the dredging activity.

Our results highlight that the outcome of concurrent pressures is not always additive, and in fact may change from antagonistic to synergistic across realised pressure fields, potentially complicating efforts to assess global cumulative impacts[6,40]. While pressures such as thermal stress associated with climate change can only be managed at a global scale, management of local pressures may, in some cases, have the capacity to modify their overall impact. A clear understanding of how locally manageable pressures interact with global and regional pressures (and via which pathways) is critical for effective management. Evidence for a shading effect associated with reduced light lends support to the idea that it may be possible to alleviate the coral loss associated with thermal stress through active mitigation efforts[41], at least at local scales.

Methods

Study details. From 19 May 2010 to 31 Oct 2011 (530 days), a large-scale capital dredging project was undertaken on the reefs around Barrow Island located ~50 km offshore the Pilbara region of NW Western Australia (Supplementary Figure 1) to provide ship access to a liquefied natural gas (LNG) processing plant on the Island. The project used a mix of trailer suction hopper, cutter suction and backhoe dredges, removing an estimated 7.6 Mm^3 of predominantly unconsolidated, undisturbed carbonate sediments overlying limestone pavement[42]. It was predicted during the environmental impact assessment processes that the dredging campaign would have some effect on corals, and these impacts were permitted to occur under state and federal approval conditions. The project was carried out under a rigorous environmental management plan[43].

The project involved collection of extensive time series of both water quality and coral health monitoring across 25 sites covering a gradient from near (< 1 km) to = far (>30 km) from the dredging activities[42]. Water quality and coral health monitoring data were collected throughout the duration of the project, which also included a period of widespread thermal bleaching which occurred in the summer of 2010–2011 due to a warm water anomaly[22,44], providing a unique opportunity to explore the cumulative impacts of suspended sediment loads and thermal stress on coral reefs. Access to associated monitoring datasets, including water quality and coral colony images throughout the dredging project were obtained through the WAMSI Dredging Science Node (https://www.wamsi.org.au/dredging-science-node).

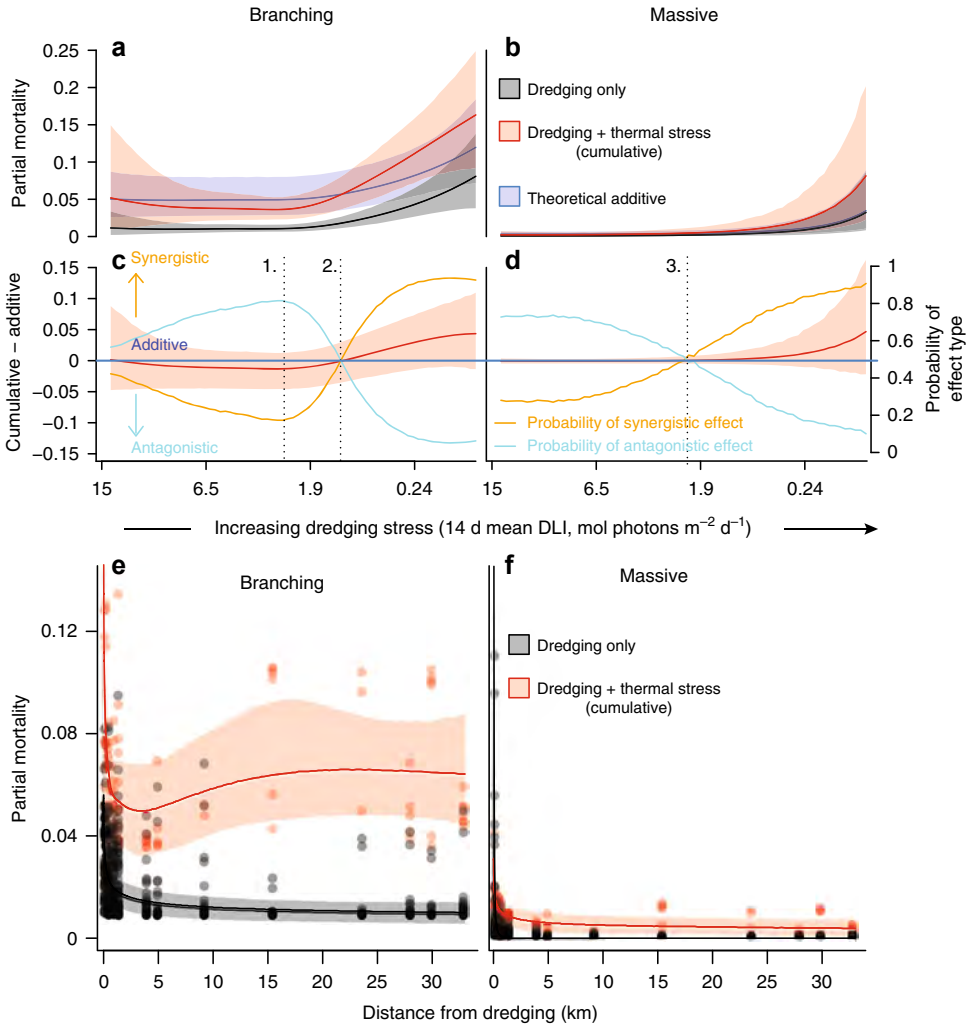

Fig. 4 Cumulative impacts of suspended sediments and thermal stress. Fortnightly partial mortality (total proportional coral loss) was calculated across a gradient of increasing dredging-related stress (decreasing 14 d running mean light values) for both branching (**a**) and massive corals (**b**). The red lines indicate the cumulative effect of dredging pressure and thermal stress, whereas the black lines indicate the effect of dredging alone. The blue line represents a theoretical "additive" curve, estimated as the sum of coral loss associated with thermal stress alone (no dredging stress and high light levels) and that predicted for different levels of dredging stress at low temperature. Shaded bands represent the 95% confidence bounds of the Monte-Carlo simulation. Additive impacts were subtracted from the cumulative impacts to determine if impacts were synergistic (greater than expected independently) or antagonistic (less than expected independently, **c**, **d**, Methods 'Estimating individual and cumulative impacts'). Fortnightly partial mortality was also examined against distance to dredging activities (**e**, **f**, Methods 'Distance of impacts') under both high (31 °C, red bands) and low (25 °C, black bands) thermal stress conditions. Three light based thresholds were derived, including: 1. the maximum probability of antagonistic impacts, 2. the intersection between antagonistic impacts and synergistic impacts for branching corals, and 3. the intersection between antagonistic impacts and synergistic impacts for massive corals

Before the start of dredging ~25 water quality and coral health (see below) monitoring sites ranging in depth from 4 to 11 m were installed from a few hundred metres to several tens of kilometres away from the future dredging activities (Supplementary Figure 1). Sites were installed from June 2009 onwards and the installations were complete by the start of the dredging on May 20, 2010. During dredging there was a near continuous unidirectional, southerly movement of the sediment plumes throughout the 1.5 y study (see ref. [45,46]). For the purposes of this study, analysis of water quality and coral health was restricted to 17 sites to the south of the main dredging and 2 reference sites (AHC and REFN) located > 28 km to north of the study area (Supplementary Figure 1). The sites were grouped into 3 categories based on previously established patterns in the severity of water quality conditions[46], including: near dredging (< 1.5 km), intermediate distances from dredging (4–15 km), and far from dredging (24–32 km), encompassing sites both to the north and south of the primary dredging site (Supplementary Figure 1).

Water quality monitoring. Water quality measurements were made at 10 min intervals for the pre-dredging and dredging phases using instruments attached to a seabed mounted steel frame. Turbidity was measured using sideways facing optical fibre backscatter (OBS) nephelometers while photosynthetically active radiation (PAR) was recorded using 2π quantum sensors. For all turbidity data, any values < 0 NTU were removed, and a smoothing filter was applied where any value > 3 NTU if the value was more than 2.5 × the mean of the preceding and following value[42]. Benthic light was measured using a 2_{Π} quantum sensor, with 10 min readings modelled using generalised additive models (GAM) for each day separately, with predicted values used to determine the sum of the per second quantum flux measurements[46,42] as the daily light integral (DLI) calculated as mol photons m^{-2} d^{-1}. The instrument platform also included a sediment accumulation sensor[47,48] which, while unable to provide absolute units of sedimentation, provides a relative sedimentation index[49], and was scaled to between 0 and 1. Water temperature (degrees Celsius, °C) was recorded by the in situ loggers. All data were visually screened for anomalies and evidence of logger failure using time series plots of the raw data, and any suspect data were removed, with all removed data recorded in a data screening log[42].

Coral health monitoring. For each of the water quality monitoring sites, 50 coral colonies considered representative of the local reef community were selected for sequential monitoring and tagged. Additional colonies were also tagged to ensure a

minimum of 20 massive Poritidae (*Porites lobata* and *Porites lutea*, hereafter referred to as *Porites* spp.) colonies per site. Each colony was photographed in plan view (from above) using a digital camera, with a set of reference photographs taken at the start of the program and used to create a reference guide showing the original image of each colony to compare against for each additional survey, to ensure that the correct corals were photographed on each occasion, and that consecutive photographs were taken at the same orientation[50]. A metal frame was used to ensure all photographs were taken at the same distance from the substratum, and also provided a scale. The colonies were photographed every ~14 d for the 530 d duration of the dredging (until 11 November 2011) with 27–40 separate field surveys undertaken to photograph each individual tagged coral colony at each site over the duration of the project.

The growth form of each colony was recorded as either encrusting, foliose, corymbose, branching or massive. Consecutive photographs of each individual colony were scored for partial mortality, cover by sediment, and thermal bleaching, as a percentage of live tissue of the specific tagged colony, based on a relative assessment of the preceding photograph(s). Percentages were recorded on a categorical scale ranging from 1 to 7 where 1 = 0%, 2 = 1–5%, 3 = 6–33%, 4 = 34–65%, 5 = 66–95%, 6 = 96–99%, and 7 = 100% coverage, which were converted to their equivalent proportional cover midpoints (1 = 0, 2 = 0.03, 3 = 0.19, 4 = 0.495, 5 = 0.805, 6 = 0.970, and 7 = 1) and used as a continuous response variable in subsequent summaries and analyses. Because of the photographic time sequence, it was possible to follow the fate of the coral tissue in time, making it possible to clearly observe mortality and/or bleaching of tissue, and in many cases identify the cause (thermal or sediment related stress). Colonies often became partially covered in sediment and if sediments were ultimately washed off the coral surface (by waves or currents) revealing live tissue, the sediment covered tissue was classified as 'live' throughout the relevant period of the photograph sequence. If the sediments remained on the surface, or once washed off revealed a dead surface, the tissue was classified as 'dead' from when the sediment covering was first observed. The same principle was used in the case of mucous sheet formation in massive *Porites* spp.[51]. Similarly, the image time series allowed bleaching associated with sediment sitting on coral tissue and being subsequently washed off to be distinguished from other, non-sediment related bleaching (largely thermal stress related).

Where it was clear from the photographic time series that a colony had been moved or dislodged either by swells and waves associated with tropical storms (TC Bianca or TC Carlos, see Supplementary Figure 2), the colonies were excluded from all further analyses.

While a broad range of coral families and genera were included across the original data set as being 'representative of the community', in most cases there was insufficient replication at each site for meaningful analyses of individual families. Instead, we focus on two key groups: 'Massive' and 'Branching', representing two contrasting growth forms that likely differ in their susceptibility to both sedimentation and light stress. Because massive *Porites* spp. were a focal group for compliance monitoring during the dredging program (see above) they occurred in sufficient numbers across the full design to be analysed as a single group (400 colonies across 17 sites, median number per site per field survey = 23). While other massive colonies were also observed (for example from the Mussidae, Diploastreidae and Lobophyllidae families) these were not generally well represented across sites, and their inclusion may bias results given that family level differences can be much stronger predictors of bleaching susceptibility than overall growth form[52]. Thus for the purposes of this study the term 'Massive' refers specifically to massive and sub-massive Poritidae, most likely a mix of *Porites lobata* and *Porites lutea* (see above). Compared to the massive Poritidae, observations of branching corals within any one family were scarce and it was necessary to aggregate the branching and corymbose *Acropora* spp. with the Pocilloporidae (all of which were branching forms, 241 colonies across 17 sites, median number per site = 10). The genus *Acropora* was the most abundant branching genera across the data set, with good spatial representation. The Pocilloporidae included representatives from *Seriatopora* spp., *Stylophora* spp. and *Pocillopora* spp., all of which have relatively similar morphologies and thermal stress sensitivity to the branching/corymbose *Acropora* spp.[38,53], and were also well distributed across the study design. Studies including species from both the *Acropora* spp. and Pocilloporidae indicate similar sediment clearance capabilities[30] and similar responses to light reduction[19]. The only other (non-Poritidae) branching/corymbose corals in the data set were a small number of *Astreopora* spp. and *Hydnophora* spp., both occurring only at two sites. These two additional branching groups were not included as they differ markedly to the branching *Acropora* spp. and Pocilloporidae in their sensitivity to bleaching[38] and because of their poor spatial representation. While there are certainly some differences in the life history characteristics of our 'Branching' group (such as mode of reproduction) that might result in different rates of recovery from disturbances[54], for the purpose of exploring the relative effects of thermal and sediment related stress on adult massive versus branching corals, members of the 'Branching' group can reasonably be considered to be functionally similar.

Predicting coral mortality. To explore the relative importance of the different suspended sediment and thermal stress pathways for mortality of corals at Barrow Island, and to build predictive models of coral mortality, a range of environmental and coral parameters were derived and used in statistical analyses (Supplementary Table 1). This included: two stress indicators derived from coral health parameters collected as part of the image analysis ('Bleached' and 'Sediment cover'); three environmental parameters based on water quality logger data designed to capture the three stress pathways associated with the suspended sediment loads cause by the dredging activities ('SSC', 'light' and 'sedimentation, see[16]) (Supplementary Table 1); temperature (also based on site specific water quality logger data); and depth and distance to the dredging activity.

All the data were processed, screened and analysed in R (version 3.2.3[55]). We used two approaches to investigate which mortality pathways have most support based on this Barrow Island data set, for both the branching and massive coral groups. In the first approach we used path analysis implemented in a structural equation modelling framework, via the sem package[56,57] in R. We used the logit of the proportional mortality of corals between each image (live cover at time t – live cover at time t + 1) as the response, for the branching and massive coral groups separately (see 'Coral health monitoring' above), averaged at the site and field survey level ($n = 473$ field survey x site means for 'Branching' and $n = 478$ field survey × site means for 'Massive'). An a priori full model was developed for each coral group (see Supplementary Figure 3). For path analyses we were explicitly interested in the relative importance of the three suspended sediment pressure pathways (SSC, light and sedimentation, see[16]) as well as thermal stress indicators (captured in our path analysis as 'bleaching' and as elevated water temperature) in causing mortality of corals. The suspended sediment concentrations (SSC) could directly cause mortality, or could cause reductions in light availability and increased sedimentation. Reduced light availability could directly cause mortality. Sedimentation could cause partial mortality directly, or via sediment cover, which may subsequently result in mortality. As branching corals were never observed with sediment covering their surfaces, this link was not included in the branching coral partial mortality pathways. The total standardized effect of SSC, light and sedimentation was calculated by multiplying the standardized coefficients within a pathway then summing the coefficient values for all relevant pathways. The total standardized effect for thermal stress was the sum of the pathway coefficients via the effect of elevated temperature on bleaching, which in turn impacted the response, as well as the direct effect of elevated temperature (see Supplementary Figure 3).

In addition to the path analysis, we used a full-subsets modelling approach[58] to determine which of the available predictor variables were most important in driving partial mortality of branching and massive corals, and to establish the best statistical models for predicting partial mortality under thermal and suspended sediment related stress. In this approach a complete model set was constructed and subsequently compared using Akaike Information Criterion for small sample sizes (AICc) and AICc weight values (ω)[59]. The relative importance of each predictor variable was determined by summing the ω values for all models containing the variable, with higher summed values representing increased importance of that predictor to the response variable[59]. To avoid issues associated with collinearity between predictors, predictors were only included in a single candidate model when their absolute Pearson correlation coefficient was less than 0.4. To ensure models remained ecologically interpretable, only models with up to three predictors were included in any one candidate model (although all predictors were examined across the whole model set). We also investigated interactions between factors (Bleached and Sediment Cover) and continuous predictors where these were deemed appropriate.

To explicitly separate bleaching impacts from mortality, the probability (or incidence) of bleaching was examined as a function of light and temperature, with morphological group (branching or massive) included as a factor. Data were the number of colonies observed during a field survey showing a higher bleaching score than the previous survey (successes in a binomial call) modelled as a function of the number of observed colonies (trials in a binomial call, median number per site per field survey = 10 for 'Branching' and 23 for 'Massive'). A site identifier was included as a random effect to accommodate repeated sampling of the same sites over time.

As the observed partial mortality events across the time series were highly zero inflated for both morphological groups, we used a two-step hurdle approach to model mortality. Firstly, the probability of observing a partial mortality event (a colony photographed during a field survey showed a higher mortality score than in the previous survey) was modelled at the individual colony level, including colony ID as a random effect to accommodate repeated sampling of the same colonies over time (median number of surveyed colonies per site per field survey, as above: 10 for 'Branching' and 23 for 'Massive'). In the next step, only the data where partial mortality events were actually observed were included, and the amount of coral loss (live cover at time t – live cover at time t + 1) modelled as a function of the proportional live cover at time t. Colony ID was again included as a random effect. Sample size for the presence only model was obviously much lower than for the presence-absence model, as this was limited only to colonies showing some partial mortality during a field survey. There were 318 and 164 colony level observations across all sites and field surveys for the 'Branching' and 'Massive' group respectively.

Models were fitted using the gamm4 function in the gamm4 package[60], with the presence of mortality modelled as a binomial distribution. We used generalised additive mixed models rather than generalised linear mixed models to allow for potential non-linear relationships between the response variable and the various continuous environmental predictors. Smoothing terms were fit using a cubic

regression spline[61], with the 'k' argument limited to 5 (to reduce over-fitting and ensure ecologically interpretable monotonic relationships). To further reduce overfitting, the maximum number of predictors allowed in any given model was restricted to three. A null model consisting of only an intercept and the random factors was also included in the model set to test if any of the included variables were indeed useful predictors of coral mortality events. While the data modelled here are time series, temporal variograms did not show strong consistent trends in temporal autocorrelation that could be modelled effectively using the correlation structures available in mgcv. Overall, temporal dependence was not strong because only a single data point from the daily water quality time series were used from the time step between observations of individual coral heads. Inclusion of colony level ID random effects effectively deals with the non-independence associated with some colonies being simply more robust or more sensitive to mortality and bleaching.

Estimating individual and cumulative impacts. We fitted the best models identified through the full subsets modelling for predicting the incidence of bleaching, the incidence of mortality and proportional coral loss (given mortality) in a Bayesian framework using the function stan_gamm4 from the rstanarm package[62] in R. During initial Bayesian fits, we used three chains with 10,000 iterations and the rstanarm default warmup of (floor(iter/2) and default uninformative priors (see[62]). As there was good chain mixing final models were fit using only one chain to obtain posterior samples for each step in the mortality pathway to use in a Monte–Carlo simulation to explore the impacts of thermal and suspended sediment related stress individually and as cumulative impacts. We focused on light as the primary suspended sediment related stressor because light was overwhelmingly the most important variable for bleaching in both groups of corals, the best predictor of mortality incidence in branching corals, but was also an important predictor of the amount of lost tissue in massive corals (Supplementary Table 2, Fig. 1). For the massive corals, where the incidence of mortality was strongly influenced by sedimentation, we estimated sedimentation pressure for each light level based on a Bayesian model fit of the observed water quality data (Supplementary Figure 5). In most cases we used the model with the lowest AICc (Supplementary Table 2) as our selected 'best' model. However, for the probability of live tissue loss in massive corals, we use the simplest model within 2 AICc (light + bleaching status), largely because this model included light rather than sedimentation, which was both more closely linked to the dredging stressor used in simulations, but also had a higher summed ωAICc importance score (Fig. 2).

For each iteration of the Monte–Carlo we predicted the incidence of bleaching, the incidence of mortality (which is dependent on bleaching status for both branching and massive corals), and the overall coral loss given mortality (also dependent on bleaching status). By multiplying the predicted probabilities for both the bleached and unbleached mortality pathways and then summing both pathways, we could calculate the total combined probability of coral loss under thermal pressure alone (high temperature and high light conditions in the absence of dredging; 31 °C and 8.8 mol photons m^{-2} d^{-1}); dredging stressors alone (25 °C, range of light levels); and under cumulative thermal and dredging stress (high temperature [31 °C] across a range of light levels). A theoretical 'additive' curve was estimated as the sum of coral loss associated with thermal stress alone and that predicted for different levels of dredging stress at low temperature.

Distance of impacts. To provide context for the spatial scale of realised impacts of suspended sediments associated with dredging in the present study, as well as the potential scale of the shading effects that actually occurred, we estimated proportional coral loss as a function of distance from the dredging activities using the actual fortnightly water quality conditions observed at Barrow Island. We used predicted rather than observed total coral loss in order to partition the effects of dredging from that of thermal stress, and examined decay relationships with distance under high (31 °C) and low (25 °C) thermal stress scenarios. The data used were the observed worst case fortnightly values for light and sedimentation (see Supplementary Table 1) at each site, reflecting the data used in the original analyses on mortality (see 'Predicting coral mortality' above). As thermal stress events primarily occur during summer months (when light levels as measured through DLI are naturally higher[46]), our thermal stress scenario was based only on the data from the months of December, January and February. The total predicted fortnightly coral loss was modelled as a binomial function of starting coral cover (100 trials) against log(distance), where distance was the minimum distance to the dredging footprints as per[46]. We modelled distance on a log scale as previous analyses had indicated that water quality conditions change as an exponential function with distance from dredging[46]. Models were fit using stan_gamm4 with default priors and warmup, with 3 chains and 10,000 iterations. Site identifier and fortnight number were included as random effects to accommodate non-independence in time and space. An estimated 10% 'Effect Distance' (ED_{50}) was calculated as the distance at which the predicted curve drops below the 50% change from the maximum predicted value (at the closest observed distance of 0.19 km) to the minimum predicted value (at the farthest observed distance of 32.8 km, except for branching corals where the minimum occurred at intermediate distances from dredging, see Fig. 4).

Reporting summary. Further information on experiment design is available in the Nature Research Reporting Summary linked to this article.

Data availability

The data that support the findings of this study are available through the Western Australian Marine Science Institution but restrictions apply to the availability of these data, which were used under license for the current study, and so are not publicly available. Data are however available from the corresponding author (R.F.) upon reasonable request and with permission of the Western Australian Marine Science Institution and Chevron Australia.

Code availability

All R code used in the analyses presented are available at: https://github.com/AIMS/WAMSI-DSN-cumulative-impacts-bleaching-and-dredging

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Author contributions

P.B-B., R.J., and R.F. conceived the study. P.B-B. analysed the images. P.B-B. and R.F. conducted the statistical analysis. R.F. led the writing of the manuscript with P.B-B and R.J.

Additional information

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RESEARCH ARTICLE

Sediment deposition and coral smothering

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Data Availability Statement: The data underlying the results presented in the study are stored at the Pawsey Supercomputing Centre (https://data.pawsey.org.au/ based in Perth, Western Australia). Permission to use the data must be sought from the data custodian, the Western Australian Marine Science Institution (www.wamsi.org.au, based in Perth, Western Australia) and are available upon entering a data sharing agreement and under the restriction that the data is used for studies that intend to 'improve the ability to predict and manage the effects of dredging'.

Abstract

Dredging in the marine environment to create and maintain safe, navigable shipping channels, and subsequent disposal of the material at sea in dredge material placement sites (spoil grounds) can generate large quantities of suspended sediment that can impact upon epibenthic marine communities. For sensitive taxa such as hard corals, understanding the mechanisms of mortality and the spatial scale over which these occur is critically important for impact prediction purposes, management of dredging using zonation schemes, and also public perception. We describe the sediment deposition field from suspended sediment falling back out of suspension created around a large (7.6 Mm^3) 1.5-year capital dredging project on a reef, using data from 2 weekly repeat observations of >500 individually tagged corals at multiple locations from 0.2–25 km from the dredging. The observations were supported by concurrent *in situ* measurements of proxy suspended sediment concentrations, underwater light, and sediment deposition (using optical backscatter sensors), and before and after surveys of seabed particle size distributions (PSDs). The distance at which 90% of the effect (from maximum to minimum) had dissipated (ED_{10}) was 20 km away from the dredging for suspended sediment concentrations (estimated via nephelometry), and underwater light (measured using PAR sensors) associated with turbid plumes, 14 km for sediment deposition (measured using optical backscatter sensors) and 4.6 km for changes seabed clay and silt content (PSD analysis). The ED_{10} for smothering of corals (the build-up of pools of loose sediment on the surface that could not be removed by self-cleaning) occurred much closer still at 3–3.3 km or (0.5–0.6 km for an ED_{50}). Smothering was common on encrusting and foliose forms where sediments accumulated in hollows and massive hemispherical forms where surface undulations (bumps) allowed sediments to pool. Smothering was never observed on branching species, even under extreme levels of sedimentation. Sediment smothering resulted in tissue bleaching and partial mortality (lesion formation), but if sediments were removed (by currents) bleached areas regained pigmentation over weeks and there was regrowth/reparation of lesions over weeks and months even before the dredging was completed. Overall sedimentation tolerance was highly related to coral morphology and surface inclination and the ability to avoid smothering by having uninterrupted downhill pathways for sediment transport across the colony.

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Introduction

Dredging in the marine environment to create and maintain navigable shipping channels and allow safe ship access, is a usual component of many large marine infrastructure developments [1]. Dredging involves the removal of sediment and/or rock from the seabed [2] and the excavation and subsequent disposal at sea in dredge material placements sites (spoil grounds) can generate large quantities of suspended sediment that can impact upon epibenthic marine communities [1, 3–5].

Well recognized cause-effect pathways include suspended sediment interfering with filtering and feeding mechanisms, increased turbidity (water cloudiness) changing light quantity and quality (for benthic primary producers), and increased sediment deposition causing smothering [5]. Predicting what may occur before a dredging program at the environmental impact assessment (EIA) stage, and managing dredging when underway through environmental monitoring, is predicated upon establishing a relationship between these dredging pressures (light reduction, suspended sediment, sediment deposition etc.) and biological responses in underlying communities i.e. developing thresholds or guidelines. In Australia, these are used in zonation schemes [6, 7] and proponents (i.e. those proposing to undertake the dredging) need to define, beforehand, zones of high and moderate impact, and also a zone of influence, which is where the plumes are likely to be seen but where there are no biological consequences. These predictions are important at the EIA stage of a project (where forecasts are made of possible environmental effects) and also in monitoring programs associated with the dredging phase, where proponents need to make sure they are compliant with the EIA predictions.

The problem is that the dredging pressures can act either individually or more likely in combination and also at different temporal and spatial scales. Knowing which is the most relevant is challenging especially because different communities (seagrasses, sponges and filter feeders, fish and corals) may each respond differently to different pressures [3, 8–10]. Managing dredging around scleractinian corals is difficult. They are phototrophic because of a mutualistic symbiosis with endosymbiotic dinoflagellates and hence are sensitive to light reduction in the water column. They can also be very sensitive to sediment deposition and have a number of different self-cleaning mechanisms to prevent sediment accumulation on their surfaces (reviewed by [11]). The process primarily involves muco-ciliary transport and corals continually shift sediments across their surfaces to prevent smothering–defined as the build-up of patches or pools of sediment that cannot be moved by self-cleaning. Smothering will reduce light availability under the sediment layer and can reduce gas (solute) exchange [12] and result in hypoxia/anoxia, and can ultimately lead to tissue necrosis (lesion formation) if the sediments remain in place and are not resuspended by waves and/or currents.

Establishing an evidence-based footprint of the scale of potential impacts associated with sediment smothering is important for dredging management. It is also important for perception of potential environmental effects associated with dredging [13]. While some turbid plumes generated by dredging can be seen for tens of kilometres [14, 15], the size of sediment deposition zones resulting in smothering of corals could be much closer to the source. Observations of corals during dredging could provide insights into these issues. Such an analysis was made possible from a major capital dredging project that occurred on the reefs in Western Australia. Since the dredging was conducted close to a diverse coral reef community there was an unprecedented level of monitoring to ensure any environmental effects complied with those approved by state and federal legislation. Over 500 individual coral heads from across multiple locations from hundreds of metres to tens of kilometres away from the dredging activity were photographed at frequent intervals for the extended duration of the project. At

each site *in situ* water quality measurements of turbidity and underwater light levels were also made before and during the dredging. Sediment deposition is particularly challenging to manage at ecologically relevant scales i.e. mg cm^{-2} day^{-1} [5, 16–19], and in the project deposition was estimated using optical back scatter techniques [17, 20, 21], in some of the first measurements of their kind for a large dredging project. Some problems were encountered with the deposition sensors, which have since been modified and redesigned as described in [22]. Although deposition could not be measured in absolute terms (as mg cm^{-2} day^{-1}), relative levels of deposition with distance from dredging could be calculated.

The images of the corals throughout the dredging program and the water quality datasets were made available by the dredging proponent for further study and analysis. Temporal and spatial patterns in turbidity and light levels during this and other capital dredging projects have been discussed in detail in [23, 24]. Patterns of coral mortality and dose response relationships have also been derived for the coral communities, with a focus on the uncertainty when combinations of stressors are involved[25]. The focus in this study is the settling of sediment and smothering of corals, the significance of sediment deposition as a cause-effect pathway and the size of the area where sediment smothering occurs compared to the long-distance movement of sediments in plumes.

Materials and methods

Water quality monitoring

Permission to undertake the dredging and to install water quality and coral health monitoring sites was given under ministerial approval statement No. 800, searchable on the Western Australia Environment Protection Authority (WA EPA) website: http://www.epa.wa.gov.au/all-ministerial-statements.

From 19 May 2010 to 31 Oct 2011 (530 days) a large-scale capital dredging project was undertaken on the reefs around Barrow Island located ~50 km off the Pilbara region of NW Western Australia (Fig 1). The dredging occurred on a 7 days × 24 h basis, with scheduled stops only for maintenance and bunkering requirements. A combination of trailing suction hopper, cutter suction and back hoe dredgers, and bed levellers were used, with dredge material placement at an offshore disposal site. The material dredged was predominantly unconsolidated, undisturbed carbonate sediments forming a thin veneer (0.5–3 m deep) overlying limestone pavements, ranging from rubble to typically gravelly sand mixed with fine silts and clays [26]. A full description of the dredging program (type and nature of the dredging and substrates) as well as data collection and methods and analyses of the water quality monitoring data can be found in [23–25].

Before the start of dredging ~25 water quality and coral health (see below) monitoring sites were installed from a few hundred metres to several tens of kilometres away from the future dredging activities (Fig 1). Sites were installed June 2009 onwards and the installations were complete by the start of the dredging on May 20, 2010. During dredging there was a near continuous unidirectional, southerly movement of the sediment plumes throughout the 1.5 y study and for the purposes of this study, analysis of water quality and coral health was restricted to 15 sites to the south of the main dredging and 2 northerly reference sites located >25 km from the excavation area (Fig 1).

Photosynthetically active radiation (PAR) and turbidity, as nephelometric turbidity units (NTU), was recorded using instruments on seabed mounted frames[23, 24]. Light data were modelled to determine the sum of the per second quantum flux measurements and the daily light integral (DLI) calculated as mol photons m^{-2} d^{-1}.

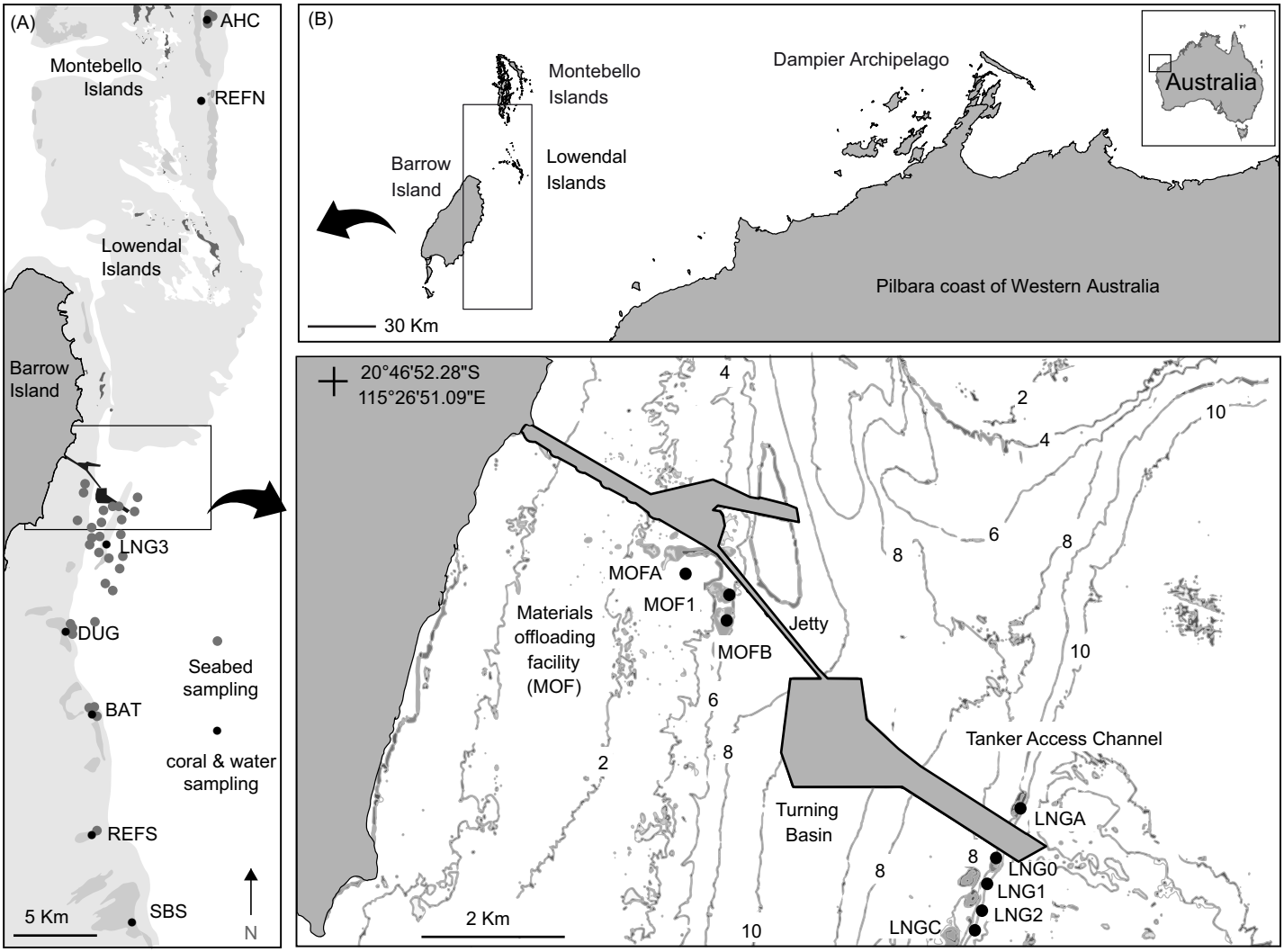

Fig 1. Location map. Location of Barrow Island in NW of Western Australia off the Pilbara coast, and the 15 water quality and coral health monitoring sites relative to the primary excavation areas (a materials offloading facility (MOF) sites and turning basin, and tanker access channel (LNG sites)).

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The instrument platform also included a sediment accumulation sensor which is based on optical backscatter sensor principle [17]. The sensor uses a light emitting diode (LED) source and fibre optic bundle set into a flat horizontal glass measuring surface. Sediment accumulating on the surface results in backscattering of light from the LED into the sensor. Every few hours a wiper removes accumulated sediment, resetting the sensor reading to zero and ideally creating a 'sawtooth' pattern in the sensor data, from which it is theoretically possible to calculate a sediment deposition rate [for a further explantion see 17, 20, 22]. In this study a sawtooth pattern was only occasionally observed and much more common were periods when sediment had obviously accumulated on the flat sensor but moved off and either resettled again or advected away from the sensor within the 2 h accumulation period. This prevented calculation of a sediment accumulation rate, but nevertheless provided information that measurable levels of sediment settling was occurring. The sensor output was averaged over a day and normalized to a scale of 0–1 for each site to produce a unit-less sediment deposition index (see [25]).

The particle size of surficial sediment samples was collected before dredging (September 2008–April 2009) and at 3 separate occasions after dredging approximately 1 year apart. Multiple scrapes of the surface 5 cm of sediment were collected by SCUBA divers within a 4 m^2 area using pre-cleaned and labelled 250 mL plastic jars. 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).

Coral health monitoring

The waters around Barrow Island support a diverse assemblage of tropical and subtropical marine fauna with over 250 species of hard corals ([27–29]). Species diversity is generally higher on the clearer water on the west coast of the island than the slightly more turbid, and lower energy waters along the eastern edge. Where the dredging took place, numerous scattered, isolated patch reefs support a varied mixed coral assemblage dominated by *Goniastrea* spp., *Porites* spp., *Euphyllia* spp., *Lobophyllia* spp., *Plesiastrea* spp., *Favia* spp., *Favites* spp., *Platygyra* spp. and *Acanthastrea* spp. and *Turbinaria* spp. and scattered hard corals such as *Acropora* spp. ([26]).

For each of the water quality monitoring sites a minimum of 50 coral colonies that were considered representative of the local reef community were tagged (marked with a unique identifier to aid future location). Additional colonies of massive Poritidae were also tagged to ensure a minimum of 20 colonies per site. Massive *Porites* spp. are difficult to identify underwater due to their small and variable corallites [30]. The species most likely included a mix of *P. lutea* and *P. lobata*, which are the most common massive *Porites* spp. in the region [28].

The colonies were photographed every ~14 d for the 530 day duration of the dredging, until 11 November 2011, and for most sites 27–40 surveys were undertaken over the dredging period. From the photographs the morphology of each colony was recorded as either encrusting, foliose, corymbose, branching or massive. Each tagged colony photograph was then scored according to the proportions of cover by sediment and for the *Porites* spp. covering by mucous sheets[31], using a categorical scale ranging from 1 to 7, where 1 = 0%, 2 = 1–5%, 3 = 6–33%, 4 = 34–65%, 5 = 66–95%, 6 = 96–99%, and 7 = 100% coverage. Where it was clear from the photographic time series that a colony had been moved or dislodged either by SCUBA divers during the photographing process or swells and waves associated with tropical storms and cyclones Bianca or Carlos, the corals were excluded from all further analyses.

Colonies often became partially covered in sediment (see below) and because of the photographic time sequence it was possible to follow the fate of the underlying tissue in time. If sediments were ultimately washed off the coral surface by waves or currents revealing live tissue, the sediment covered tissues was classified as 'live' through the photograph sequence. If the sediments remained on the surface, or once washed off revealed a dead surface, the tissue was classified as 'dead' from when the sediment smothering was first observed. The same principle was used in the case of mucous sheet formation in massive *Porites* spp. ([31]). The number of observations of sediment cover and mucous sheet formation (*Porites* spp. only) were determined and summarised for each site.

Statistical analyses

Light, suspended sediment concentration (SSC) and deposition at different distances from the dredging activities was summarised as fortnightly maximum 14 d running mean DLI, 14 d running mean nephelometrically-derived SSC (using a nephelometric turbidity unit (NTU) to SSC conversion factor of 1.8 (see [23, 24]), and 60 d running mean sediment deposition index. Running mean time scales were selected as those best predicting coral mortality [25].

Distance decay relationships were fitted for these water quality parameters using mixed model regression, with distance from dredging activity as the predictor, and fortnight number and site code included as random effects. Distance was modelled on a log scale, as previous analysis had found an exponential decay in water quality conditions with distance from dredging. Appropriate transformations were applied to linearize the relationship with $\log_{10}$(distance) and to ensure normality. This included modelling the $\log_{10}$ of NTU, the square-root of DLI and the logit of sediment deposition, which was an index between 0 and 1.

Sediment particle size distribution was expressed as relative percentage of particle sizes in each of the four classes gravel (>10–2 mm), sand (2000–62.5 μm), silt (62.5–4 μm), and clay (<4 μm), and a distance decay relationship fitted using the logit of the proportion in the combined silt and clay size class against $\log_{10}$ distance as above.

The percentage of colonies having a mucous covering or sediment smothering score of 3 or greater i.e. >5% mucous or sediment cover was also modelled on a logit scale as a function of $\log_{10}$ distance, as was the probability of non-zero mortality of *Porites* spp. (as reported in[25]).

From the distance decay relationships effect distances were calculated 50% (ED_{50}) and 10% (ED_{10}) from the predicted value at the farther site distance (34.8 km) to that of the predicted value at the closest site distance (190 m). All relationships were fit as Bayesian models using Stan and the rstanarm package package in R [32] with uninformative priors. Estimated 95% credible bands were calculated from five Markov chain Monte Carlo chains (10,000 burn-in, 20,000 iterations).

Results

Observations from the coral colony monitoring program

There was clear evidence of sediment smothering associated with the dredging program at sites very close to the dredging activities, with patches of loose, unconsolidated, fine sediment collecting on the surfaces of corals near the dredging activities. This sediment 'smothering' was most commonly observed in flattened species such as *Montipora* and *Podabacia* spp. corals (Fig 2A and 2B) and in foliose *Turbinaria* spp. corals (Fig 2C). Sediments typically became trapped in concave depressions or 'hollows' on the surfaces, sometimes building up deposits several millimetres thick. Sediment build up was also frequently observed on the uppermost section of some massive *Porites* spp., especially those with more rugose, bumpy surface morphologies where sediments accumulated in valleys between neighbouring bumps or protrusions (Fig 2D, see below). Occasionally small (mm sized) patches of sediments were observed on smooth hemispherical or rounded massive colonies such as *Lobophyllia* spp. and *Diploastrea* spp. (see Fig 2E and 2F); however, these morphologies were typically very capable of rejecting sediments even during periods of extreme levels of deposition resulting in smothering of surrounding surfaces (see background of Fig 2F).

The relative sediment clearing abilities across a range of morphologies is also indicated in Fig 3A, showing a cluster of branching, hemispherical and encrusting colony morphologies before and then near the end of the dredging program. Extensive smothering of the low relief, encrusting *Montipora* spp. has occurred whilst the overlying tabulate *Acropora* spp. and neighbouring hemispherical *Lobophyllia* and *Diploastrea* spp. show little or no evidence of smothering. Fig 3B–3D show branching *Pocillopora* and *Acropora* spp. colonies overlying massive, hemispherical *Porites* spp. colonies showing smothering of the *Porites* spp. but no accumulation of sediment on the branching forms. Fig 4 shows an extended ~360 d sequence of images of the two colonies in Fig 3C, showing no evidence of sediment accumulation on the tabulate *Acropora* spp. despite near complete smothering of the underlying *Porites* spp.

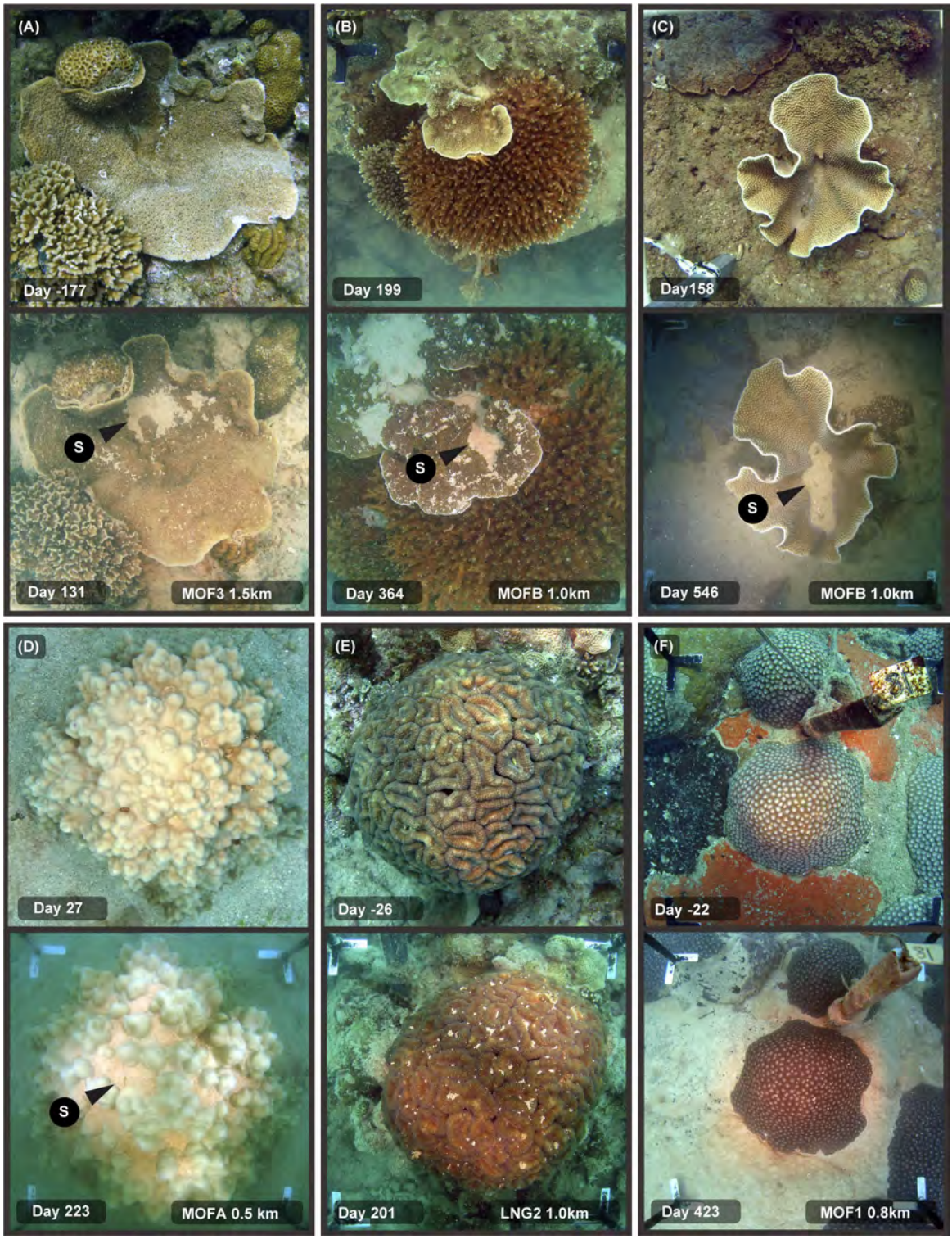

Fig 2. Sediment smothering in different coral morphologies during a large-scale capital dredging project–see text for explanation. Legends on the images indicate the site name (see Fig 1), distance from dredging activities (km), and days since the start of dredging (19 May 2010). The images show the differing susceptibility of a range of morphologies to sediment deposition with colonies with concave areas and depressions on the surface more commonly accumulating sediment compared to more hemispherical, rounded morphologies. For the arrows, S = sediment accumulation.

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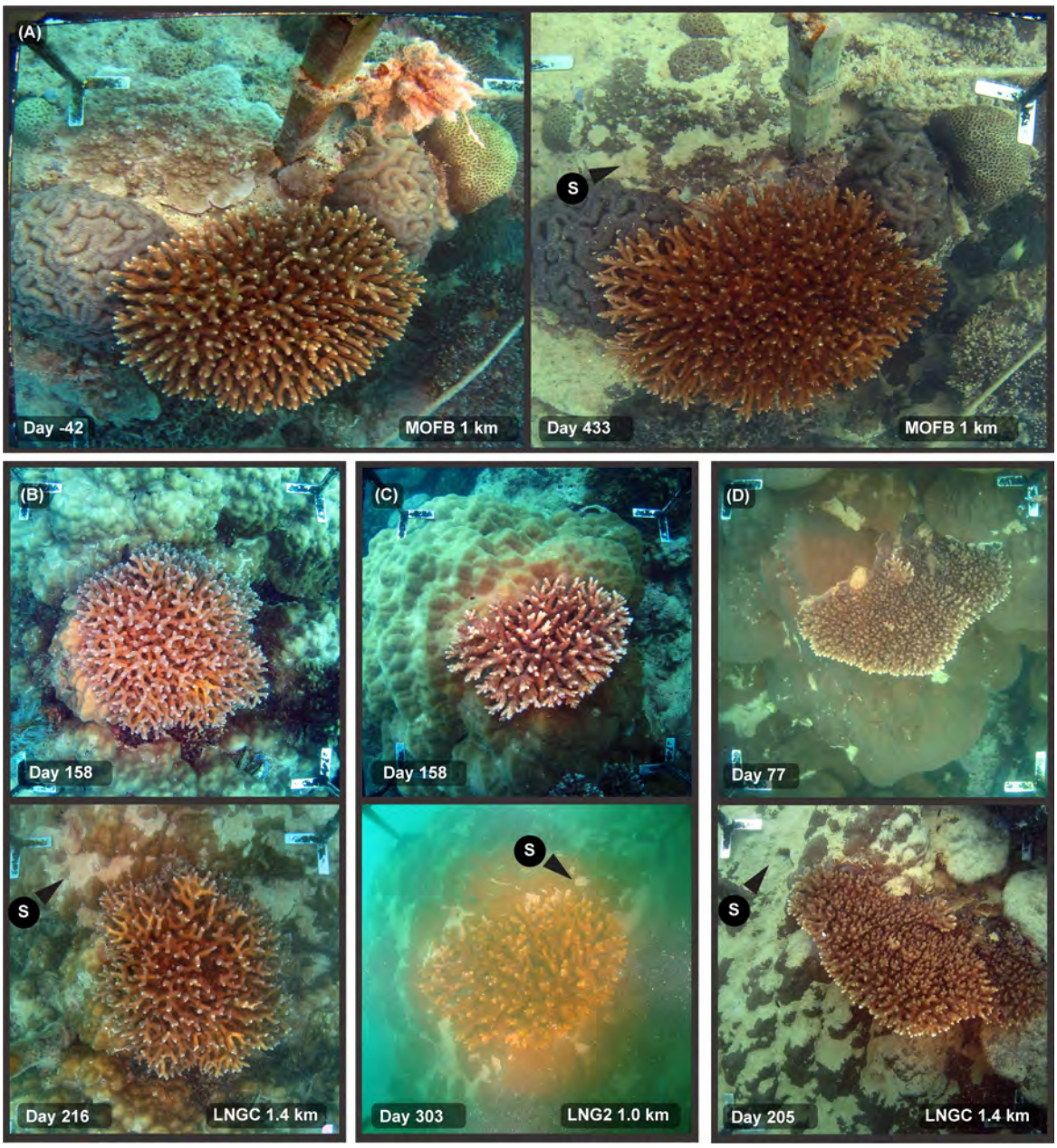

Fig 3. Sediment smothering during a dredging project, showing different susceptibility of different morphologies–see text for explanation. Legends on the images indicate the site name (see Fig 1), distance from dredging activities (km), and days since the start of dredging (19 May 2010). (A) A mix of morphologies including massive (hemispherical) colonies, encrusting and branching (tabulate) morphologies showing smothering of the flat, low lying encrusting form at Day 433. (B–D) Images showing the sediment clearing abilities of branching morphology compared to underlying massive, hemispherical morphology of the *Porites* spp. For the arrows, M = mucus or mucous sheet, B = bleaching, S = sediment accumulation, R = reparation/repair.

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Massive, hemispherical *Porites* spp. with smoother surfaces were more efficient at sediment rejection than more rugose, bumpy forms and Fig 5 contrast the fate of two neighbouring *Porites* spp. over the 530 d of the dredging. Surface smothering was regularly observed in the bumpy form were sediments became trapped in valleys as opposed to the smoother form (cf day 129, 304, 341, 407 in Fig 5A and 5B).

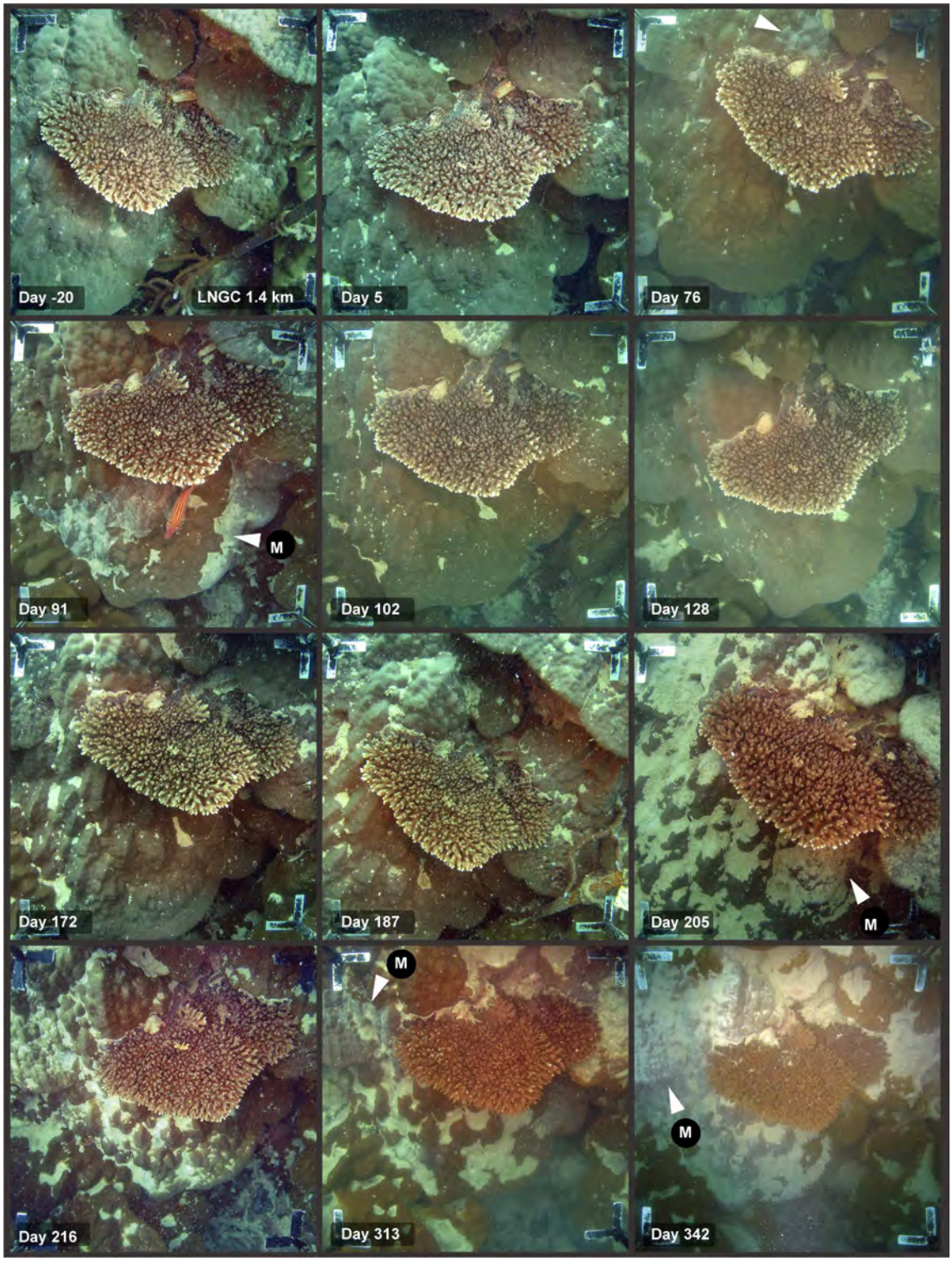

Fig 4. Time series of images showing sediment smothering during a dredging project, showing the different susceptibility of two different morphologies–see text for explanation. Legends on the images indicate the site name (see Fig 1), distance from dredging activities (km), and days from the start of dredging (19 May 2010). The images show the superior sediment clearing abilities of branching (tabulate) morphology of the *Acropora* spp. compared to underlying massive, hemispherical morphology of the *Porites* spp. which becomes smothered with fine sediments. White arrows indicate mucous sheet formation (see text).

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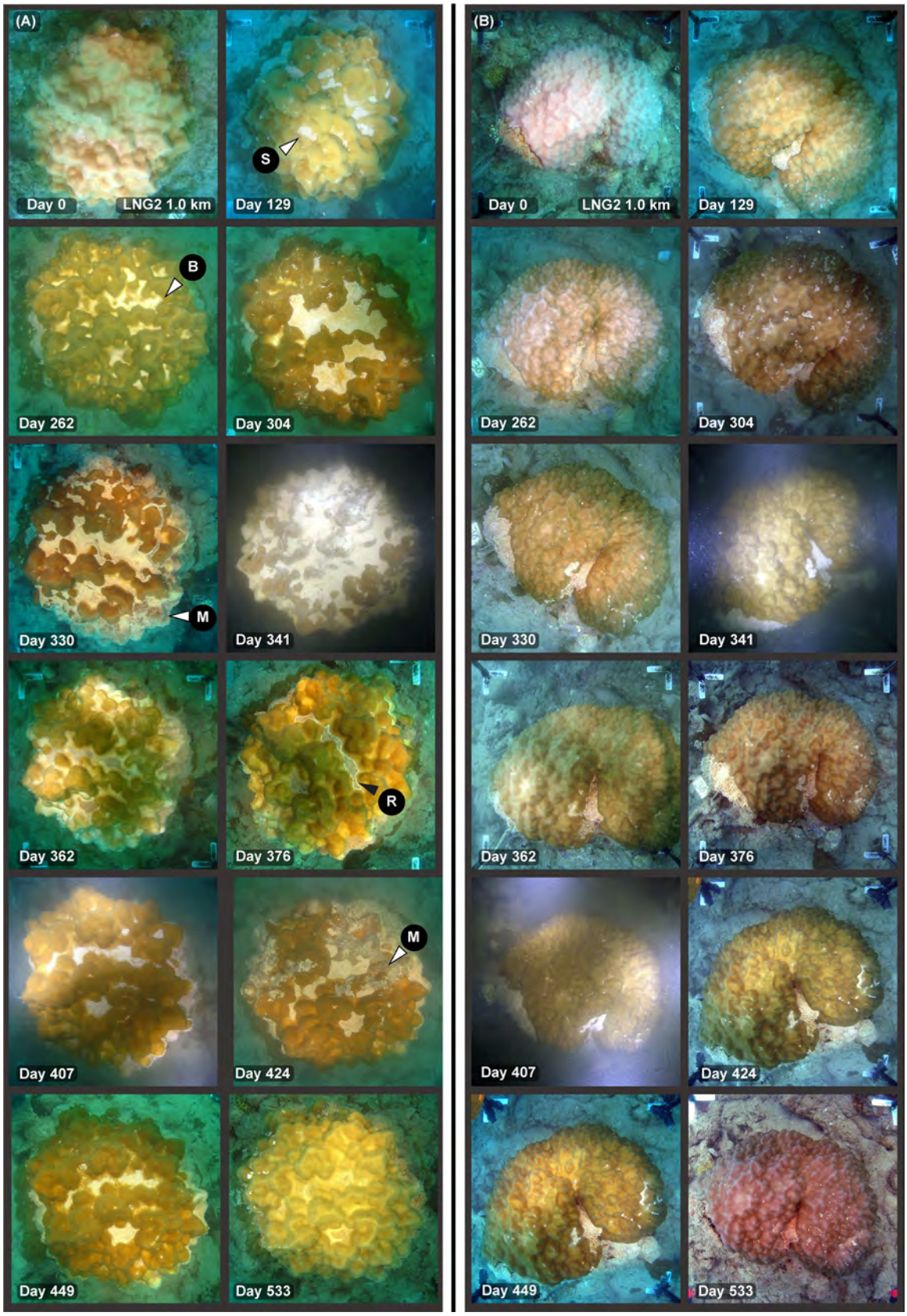
(A)
Day 0
LNG2 1.0 km
Day 129
S
Day 262
B
Day 304
Day 330
M
Day 341
Day 362
Day 376
R
Day 407
Day 424
M
Day 449
Day 533
(B)
Day 0
LNG2 1.0 km
Day 129
Day 262
Day 304
Day 330
Day 341
Day 362
Day 376
Day 407
Day 424
Day 449
Day 533

Fig 5. Time series of images showing sediment smothering during a dredging project, showing the different susceptibility of different morphologies in *Porites* spp.–see text for explanation. Colonies in (A) and (B) were from site LNG2, 1.0 km away from the dredging. Legends on the images indicate days since the start of dredging on 19 May 2010 (see text). The images show the contrasting the sediment clearing capacity of the *Porites* spp. colonies with subtly differing morphologies (bumpy versus smooth, see text). For the arrows, M = mucus or mucous sheet, B = bleaching, S = sediment accumulation, R = reparation/repair contrasting.

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Using the time series of photographs, it was possible to follow the fate of colonies once smothered. If sediments that had accumulated in hollows on the surface was re-suspended from the colony (by waves and/or currents), the tissues underneath was often discoloured or 'bleached' (see Fig 6A–6I). Bleached areas often regained their pigmentation (see Fig 6G, 6H and 6I). Fig 6J–6I shows normally-pigmented branching *Acropora*, *Pocillopora* spp. colonies overlying massive *Porites* spp. colonies where there is an unusual variegated appearance characterized by bleaching of the hollows and valleys between bumps. The pied appearance is interpreted as being caused by prior smothering by sediment based on the similarity to the pattering seen in Fig 6J–6I.

The longer time sequence in Fig 5A, also shows bleaching of areas (see arrow, day 262) where sediments have been re-suspended off the coral's surface. This is likely to be have been caused by waves and swell associated with Tropical Cyclone Bianca, which passed by Barrow Island as a category 2/3 cyclone, 105 nm away from the island at its closest point, although no effects of cyclones Bianca or Carlos on hard coral cover on the reefs around Barrow Island was reported by [33]. More sediment was observed accumulated in the hollows on days 304, 330 and especially on day 341 where the coral's surface is approximately 50% smothered by sediment (Fig 5). When next observed on day 362 the sediment had again been re-suspended from the surface showing more bleached areas (Fig 5). These bleached areas quite rapidly regained their pigmentation (over a period of 2 weeks) and our interpretation of the white leading edge of the coral tissue (see arrow in Fig 5A at Day 376) is that the coral is repairing or re growing over a central lesion in the deepest part of the hollow. After several more photographs showing sediment smothering in the hollows, the colony at the end of the dredging has only a few small patches of sediments which may be lying on a lesion or bleached tissue (see arrow in Fig 5A at day 533).

Similar patterns of smothering, resuspension, bleaching and recovery/repair were observed in *Turbinaria* spp. with an 'open' foliose morphology (Fig 7). In this colony a central lesion formed before the dredging started, and over the duration was seen with either a high sediment content filling the central lesion (Fig 7B, 7D, 7F, 7H, 7J and 7L) or with a low sediment content (Fig 7A, 7C, 7E, 7G, 7I and 7K). The photograph on day 275 (19 February 2010) was after Cyclone Bianca, and the central hollow was virtually sediment-free, with a white growing edge around a central lesion (see arrow Fig 7).

One of the most conspicuous physiological responses of the *Porites* spp. to the dredging activities was the production of mucous sheets (see [31]). These sheets, seen forming in Fig 8A, sometimes enveloped large parts or even the whole colony (Fig 8B–8D and 8G). Once formed (or forming) the sheets became fouled with sediments becoming progressively opaque. The mucous sheets eventually slough from the surface (Fig 8F and 8I) revealing clean, largely sediment-free surfaces underneath (Fig 8E, 8F, 8H and 8I). Bleaching was occasionally observed on the corals surface once the sheet had been sloughed (Fig 8H).

Some colonies were observed forming multiple sheets over the dredging project (see white arrows in Fig 4, Fig 5, and Fig 3 of [31]. Mucous sheet formation was also occasionally seen on branching *Porites* spp. (see Fig 8J). Mucus production was observed in some foliose *Turbinaria* spp. but presented as strands or web-like meshes that gathered sediments and were presumably sloughed or gathered in the central hollow (see Fig 7G, 7J and 7L and Fig 8H and 8I).

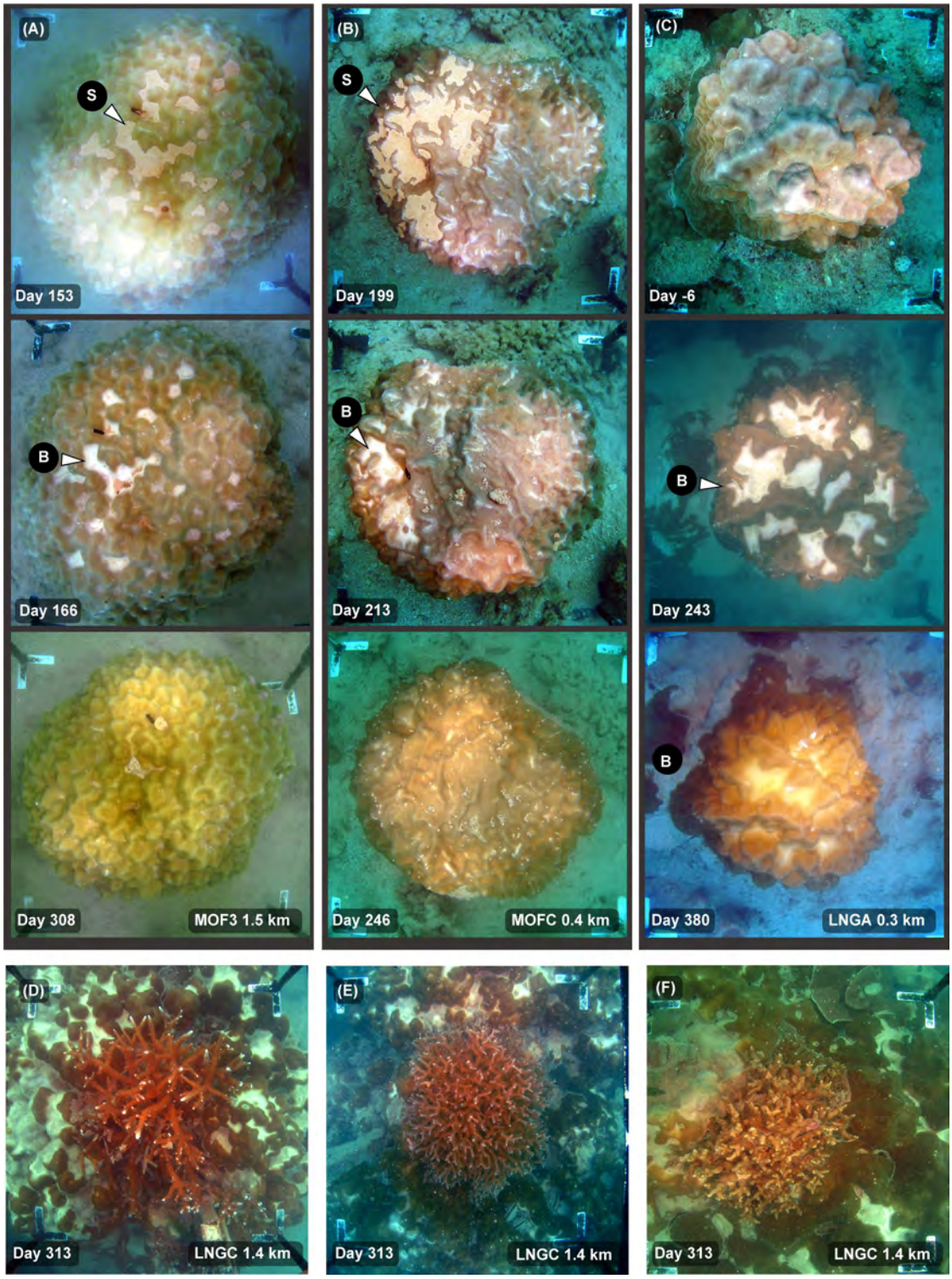

Fig 6. Effects of long-term sediment smothering on corals during a dredging project–see text for explanation. Legends on the images indicate the site name (see Fig 1), distance from dredging activities (km), and days since the start of dredging on 19 May 2010). The time series for colonies (from three different sites) over 380 days of the dredging show tissue bleaching in surface hollows of massive *Porites* spp. that were at some stage covered with sediment and eventual recovery of pigmentation. (D-F) Show similar sediment smothering and/or bleaching in hollows of the massive *Porites* spp. (background) whilst the overlying branching *Acropora*, *Pocillopora* and *Porites* species in the foreground show no smothering. For the arrows, M = mucus or mucous sheet, B = bleaching, S = sediment accumulation, R = reparation/repair.

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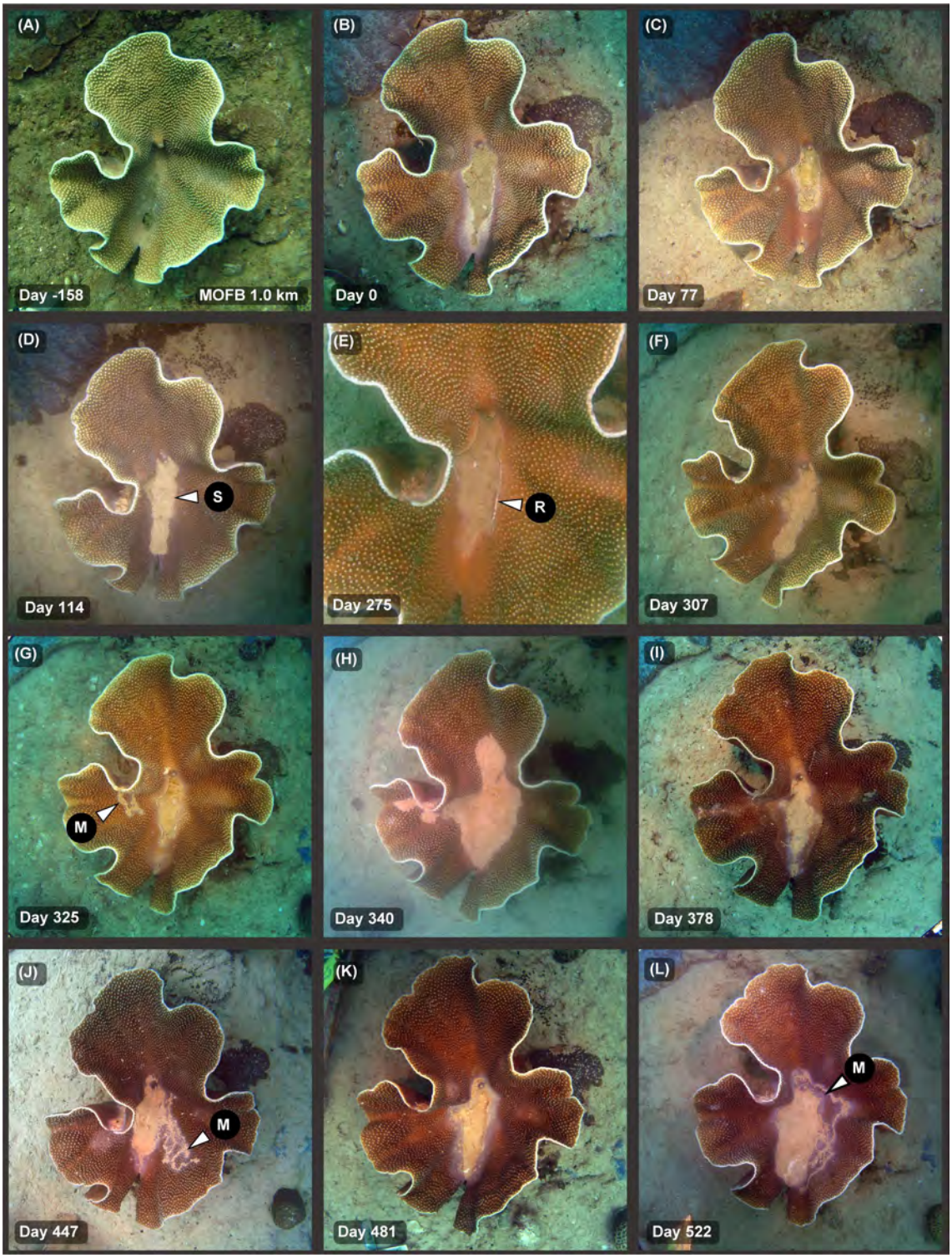

Fig 7. Time series of images of sediment smothering in a foliose *Turbinaria* spp. during a dredging project–see text for explanation. Shown is a coral colony located at site MOFB (see Fig 1) 1.0 km away from the nearest dredging over the 533 d Barrow Island capital dredging program, showing various degrees of smothering, bleaching and recovery/repair (see text). For the arrows, M = mucus or mucous sheet, B = bleaching, S = sediment accumulation, R = reparation/repair.

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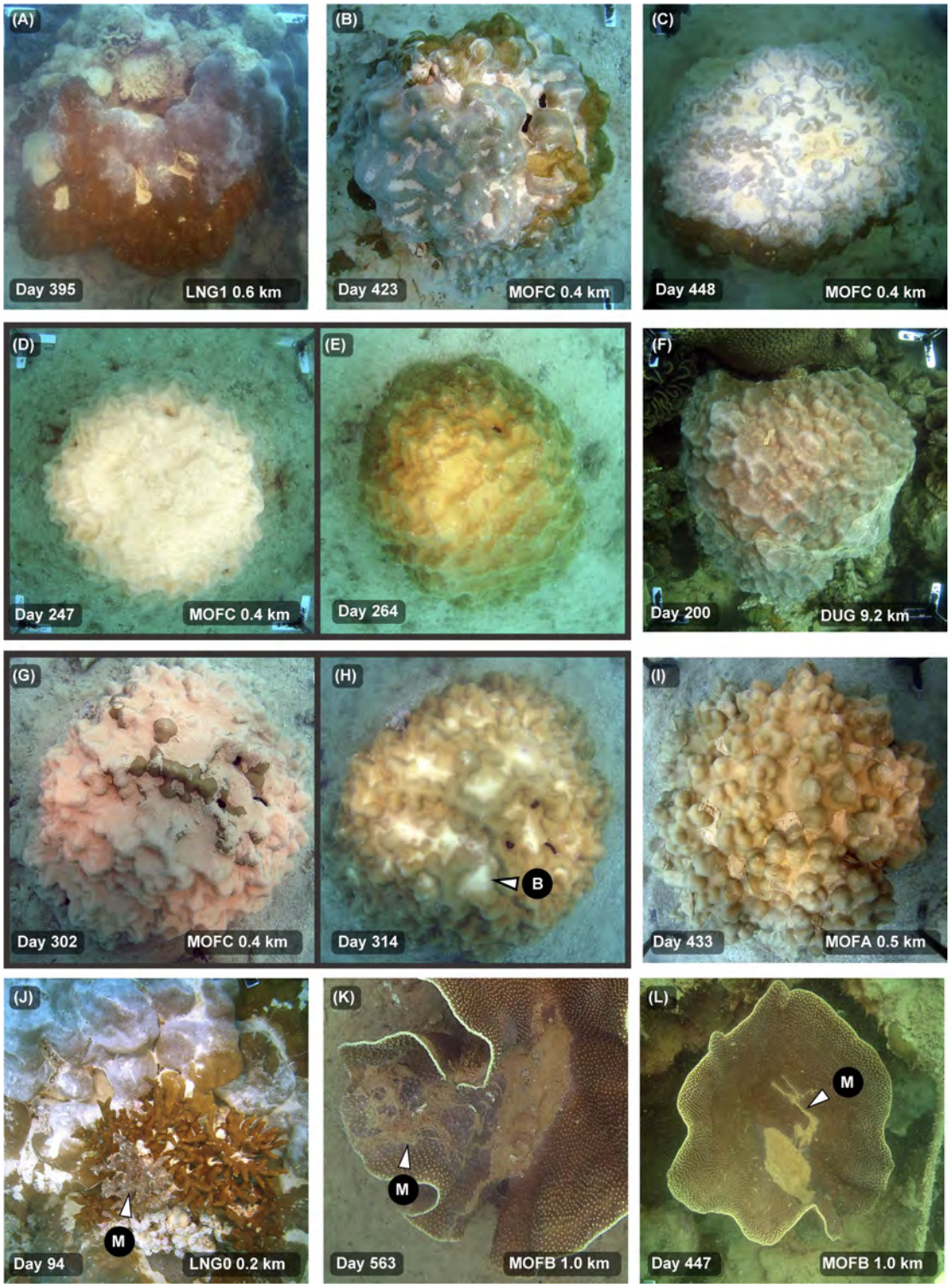

Fig 8. Mucous sheet formation and mucus production in massive, foliose and branching coral colonies during a dredging project–see text for explanation. Legends on the images indicate the site name (see Fig 1), distance from dredging (km), and days since the start of dredging (19 May 2010). For the arrows, M = mucus or mucous sheet, B = bleaching.

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Fig 9 shows evidence of growth of branching and encrusting species during the dredging program.

Water quality monitoring

Spatial and temporal patterns in water quality (turbidity and underwater light) associated with the dredging have previously been described in detail[23, 24]. Briefly, nephelometrically-derived SSCs as well as sedimentation increased markedly from the start of the dredging campaign (at Day 0) compared to the baseline-pre-dredging phase (Fig 10A & 10B). There was a strong relationship with distance from the dredging activity for all three water quality parameters, allowing calculation of an effect distance (ED) (Fig 9). Estimated distance of 10% effect (ED_{10}) for SSC (NTU) was at 20 Km (Fig 10C, Table 1). Associated with the high SSCs (hence turbidity), the ED_{10} for 14 day running mean light (as DLIs) was estimated at 22 km (Fig 10B and 10E, Table 1). Distance of observed sedimentation were slightly less than for turbidity and light, with estimated ED_{10} at around 14 km from the source of dredging (Table 1, Fig 10).

Before the dredging the particle size distributions of surficial sediments collected from 200 m to 35 km away from the dredging were predominantly sand sized (Fig 11A). In the first survey after the dredging, there was a marked increase in the clay and silt (0.2–62.5 μm) composition, seen as an upward movement of the grey circles in Fig 11A. When expressed as a change from the baseline period, the combined clay and silt fraction showed a clear distance–decay relationship and the estimated distance of effect (ED_{10}) was 4.6 km (Fig 9F). On a percentage composition basis, and for samples within 5 km from the dredging, the combined silt and clay content was 5 times higher after the dredging (Fig 11B), and although halving in subsequent surveys was still ~2.6 times the pre-dredging value 3 years after the dredging (Fig 11B).

Patterns of sediment smothering and mucus sheet production in *Porites* spp.

Over the 553 d of dredging 660 individually tagged coral colonies were examined with 6,920 (i.e. colonies × time) observations of *Acropora* spp. and Pocilloporidae colonies (branching/corymbose morphologies), and 10,821 observations of massive *Porites* spp. colonies. As discussed previously, there were no examples of sediment smothering in the *Acropora* spp. and Pocilloporidae colonies (branching/corymbose morphologies, see also Fig 3).

The percentage of observations of massive *Porites* spp. corals with >5% mucous cover (category 3 and above) ranged from ~5–10% at the sites <1.5 km from the dredging to <2% at sites ≥~5 km away (Fig 10G). The estimated distance of 10% effect (ED_{10}) for mucous sheet formation (6–33% of the colony surface) was at 3 km (range 2–4.5 km, Table 1). Patterns of sediment smothering on *Porites* spp. were very similar to those of mucous sheet formation, with the number of observations of corals with >5% mucous sheet cover (category 3 and above) ranging from ~10–40% at the sites <1.5 km from the dredging to <2% at sites >5 km away (Fig 10H). The estimated distance of 10% effect (ED_{10}) for sediment smothering (6–33% of the colony surface) was only slightly greater than that observed for mucous sheet formation, at 3.3 km (range 2.4–4.3 km, Table 1) and showed a very similar pattern to that for estimated probabilities of non-zero mortality (Fig 10H).

Discussion

Sediment deposition resulting in smothering of corals leading to mortality is a significant cause-effect pathway associated with dredging and turbidity generating activities near coral reefs. In comparing the spatial patterns of turbidity and light attenuation, seabed particle size distribution and sediment deposition, with biological responses including sediment

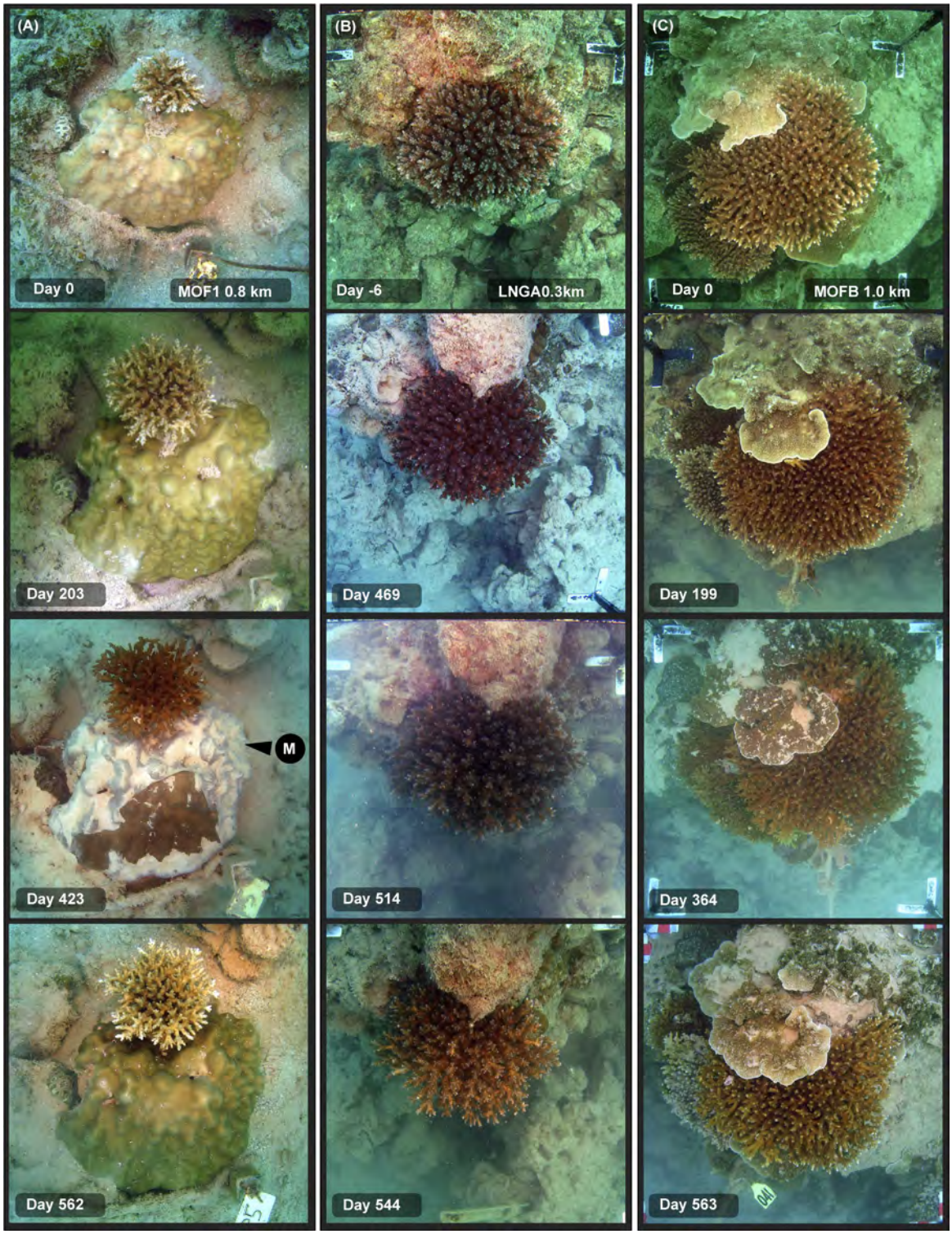

Fig 9. A, B, C. Time sequence of branching, massive and encrusting coral colonies during a dredging project–see text for explanation. Legends on the images indicate the site name (see Fig 1), distance from dredging activities (km), and days since the start of dredging on 19 May 2010). Images show the growth of some colonies over the dredging program. For the arrows, M = mucus or mucous sheet.

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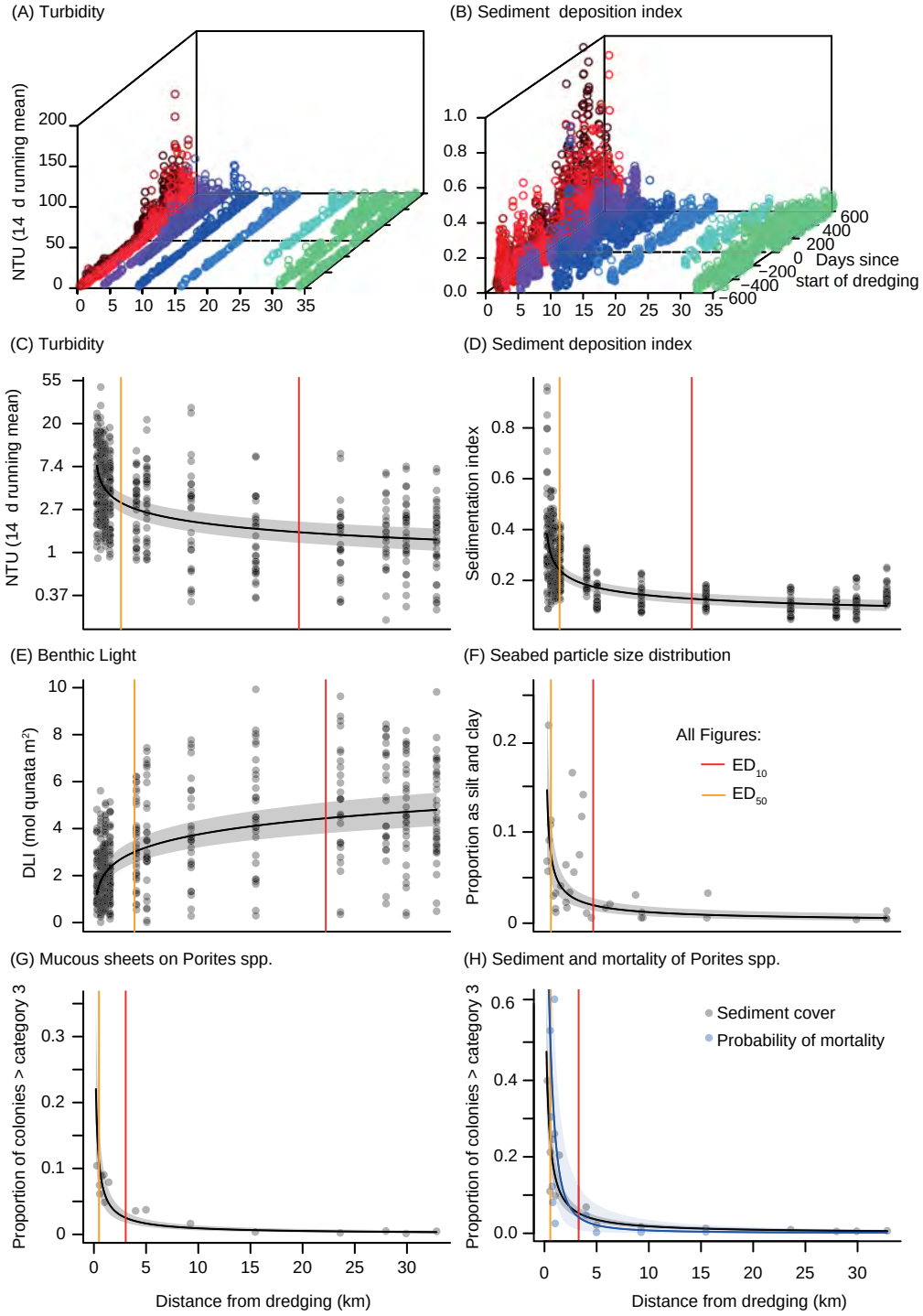

Fig 10. A-D. Distance decay relationship for physical parameters (turbidity sedimentation, light availability) and biological parameters (mucus sheet production, smothering and mortality) during a large-scale dredging project. Shown are (A) the average daily nephelometry-derived suspended sediment concentration (SSC) and (B) Sediment deposition index (range 0–1), against distance (km) from dredging (see Fig 1) during the pre-dredging, baseline phase, and from the start of dredging (Day 0, 19 May 2010) until completion (4 Nov 2011). The 2D plots show (C) fortnightly maximum 14 d running mean NTU, (D) 60 d running mean sediment deposition index, (E) 14 day running mean daily light integral over the dredging phase, the proportional increase in clay and silt in benthic sediments (F), the proportion of coral colonies having a mucous score of >3 (6–33% coverage) (G), the proportion of coral colonies having a sediment cover score of >3 (6–33% coverage) and estimated probability of non-zero *Porites* spp. mortality (H). Red and orange lines indicate the estimated distance of 10% (ED_{10}) and 50% (ED_{50}) effect as determined by fitted regression.

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Table 1. Effect distances (ED) in km for a 90% decline (ED_{10}) and 50% decline (ED_{50}) in physical pressure parameter: SSCs (NTU), benthic light (DLI), sediment deposition and changes in silt and clay content and (B) biological response parameters including: sediment smothering and mucous sheet covering in massive *Porites* spp. The physical and biological parameters were measured at 17 locations at different distances from the dredging activities (Fig 1). Effect distances were calculated from the predicted value at the farther site distance (34.8 km) to that of the predicted value at the closest site distance (190 m), with confidence bounds estimated from a posterior sample from Bayesian model fits.

	ED_{10} (km)	95% CI (km)	ED_{50} (km)	95% CI (km)
(A) Physical pressure parameters				
Nephelometry-derived SSCs	20	9.7–33	2.5	1.2–4.9
Daily Light Integrals	22	11–33	3.8	1.9–7.6
Sediment Deposition Index	14	8.3–28	1.4	0.83–2.2
Particle Size Distribution	4.6	2.7–8.6	0.56	0.19–1.0
(B) Biological response parameters				
Mucus cover >6–33% (Category 3)	3.0	2.0–4.5	0.49	0.21–0.81
Sediment cover >6–33% (Category 3)	3.3	2.4–4.3	0.56	0.34–0.81

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smothering and mucous sheet formation in corals, the sediment deposition field was approximately an order of magnitude less than the distances travelled by the plumes. Coral mortality occurred within a few kilometres south of the dredging and within the area defined by the deposition field. Establishing evidence-based footprints of the scale of potential impacts associated with sediment smothering as compared to elevated turbidity and dredging related plumes is important for dredging management, and also for perception by the public and regulators of potential environmental effects [13]. Equating risk based solely on the extent of visible plumes could be very misleading as to the overall spatial effects of dredging projects.

Spatial effects

A previous study of the turbid plumes associated with this dredging project showed they could be identified from satellite images up to 30 km away from site of excavation [34]. Analyses of

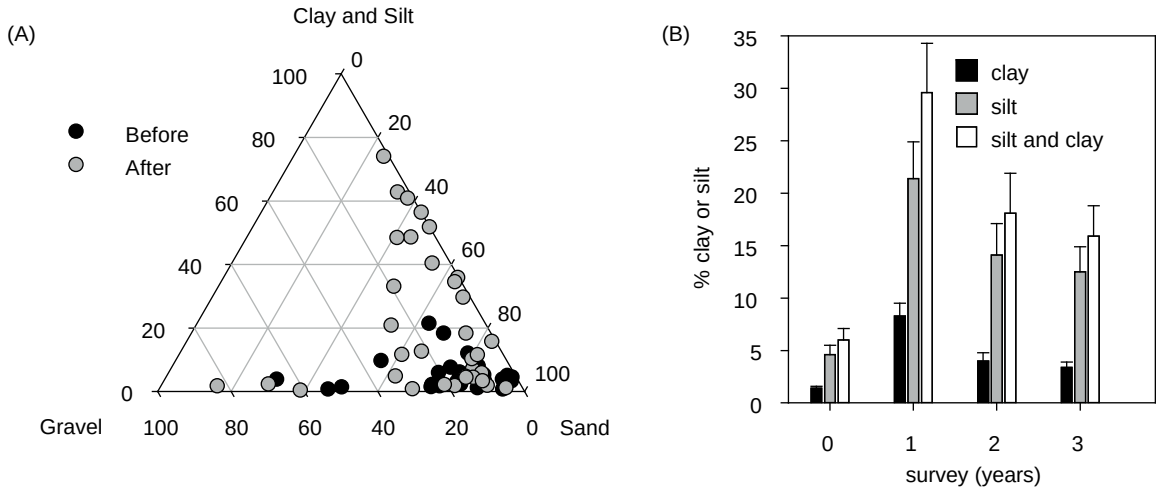

Fig 11. Changes in the particle size distribution of surficial seabed sediments collected from multiple locations from 200 m to 34 km away from the dredging before and after dredging. Shown are A) ternary plot of the percentage composition of the sediments collected (see Fig 1) expressed as percentage gravel, sand and silt and clay (62.5–0.02 μm). (B) Site level summaries of the clay and silt content immediately after dredging compared to before dredging, against distance (km) from the dredging.

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water quality data showed elevated SSCs and reduced light availability (identified by a median value above the 80th percentile of the baseline, pre-dredging data) and occurred 17–22 km from the site of dredging [23]. In this study, a different approach (the ED_{10}) was used to contextualize the spatial effects for a range of sediment-related metrics. For turbidity and light the distance from dredging at which 90% of the effect–from maximum to minimum–had dissipated was in the order of 20 km, similar to the estimates of [23]. The ED_{10} from the sediment deposition sensors was closer at 14 km. Measurable changes in the seabed clay and silt content occurred much closer to dredging with an ED_{10} of 4.6 km. For biological effects such as smothering of corals and mucous sheet formation (see below) ED_{10} values were even closer to the dredging, at around 3–3.3 km. More substantial impacts, such as a 50% effect on coral colony parameters (ED_{50}, Table 1), occurred still closer to the dredging activity, at >0.5–0.6 km.

The distance decay relationship for elevated SSCs and a reduction in benthic light [23] and sediment deposition shown in this study gives considerable authority to the use of zonation schemes (see [6, 7] as frameworks to manage dredging projects near sensitive marine environments. Clearly, spatial effects will be highly project specific and for contextual purposes this was a very large scale (7.6 Mm^3), 1.5 year capital dredging project with marked effects on water quality. The local oceanography was also unusual and characterized by unidirectional southerly flow over the project duration [23, 34].

The sediment deposition hazard associated with dredging activities arises because sediments are mobilized from the seabed into a typically low energy water column, where turbulence and hydrodynamic forces are usually insufficient to keep them in suspension. This results in elevated rates of deposition and the creation of a deposition field that can lead to smothering of sensitive organism such as corals when the self-cleaning capabilities are exceeded [5]. In this study the resettlement of the suspended sediments resulted in a 5 times increase in the silt and clay content of the seabed after dredging. Such an increase has been reported previously in dredging projects [5], but in this study the particle size distribution analyses showed increased silt/clay was still 2.5 times higher than the pre-dredging level 3 years after the dredging activities were completed. Increased siltification and the potential for increased turbidity by wind and wave resuspension is a long-term environmental legacy of dredging projects and the ecological consequences to nearby benthic communities have yet to be properly understood.

The response of coral colonies to high sediment loads varied substantially across coral taxa, as well as across different growth forms even within taxa. Smothering was very morphology-specific and common on encrusting, foliose and certain massive morphologies, but was never observed on branching species *Acropora*, *Pocillopora* and *Porites* spp. species at any stage throughout the dredging program. In morphologies where smothering occurred, sediments were commonly seen accumulating in concave depressions or 'hollows' (*sensu* [35]) on the surface, as has been reported previously [11, 36, 37]. This was particularly evident for the massive *Porites* spp. colonies where sediments accumulated on more rugose, bumpy morphologies than those with smoother surfaces (cf Fig 5) as a result of sediments becoming trapped in local minima.

Sediment clearance model for corals

Corals clear their surfaces using a range of different active (energy-requiring) mechanisms [11, 36, 38], of which the muco-ciliary transport process [35, 39] is the most important. Passive processes are also involved which are primarily associated with gravitational forces related to surface inclination and water movement [11, 38, 40, 41]. Some corals have been referred to as 'passive shedders' and using passive processes to remove sediments or using active or passive

sediment removal mechanisms [4]. However, recently it was shown that accumulation rate of fine silt and clay was only marginally lower on dead branching skeletons of a range of morphologies than on flat, two dimensional coral surfaces [42]. The implication is that sediments are not being passively shed and it seems unlikely that passive process alone can clear sediment from corals as sediments also become attached or 'embedded' [39] in the surface mucus layer of corals on initial contact. Perhaps passive shedding of sediments (not involving active processes) can occur if sands as opposed to silt and clays are used [5], and applied at high deposition rates and administered to corals briefly over several minutes.

Where passive processes play a significant role is the interaction with the active mechanisms of ciliary transport. From the many time series observations of sediment smothering over the dredging period and based on laboratory-based studies with the same or similar species and morphologies, it seems that sediment shedding ability (and hence tolerance), is simply related to the opportunity for sediments to be moved in an uninterrupted downward direction. Sediments will ultimately be shed from the colony when an edge or the base of the colony is encountered. Several earlier studies have suggested that there is typically no co-ordinated movement or 'routing' of sediments to the edges of corals for shedding, and our observations largely support this. The sediment movement process across corals has been described as a random walk [43]. However, the images collected during the dredging project clearly show that sediment transport is highly dependent on surface orientation. Experimentally it has also been shown that sediments are shed faster over descending slopes with a rate related to the angle [41, 44, 45]. By corollary, transport is slower on upward slopes. We suggest that sediment shedding in corals is via ciliary movement and mucociliary transport and is an active process and an indiscriminate random walk, but sediment transport will inevitably be quicker on downhill slopes and quicker under increased flow [38, 42]. If the transport process encounters an edge the sediments will be shed or if encountering an upward slope, it is slowed (or potentially stops if the slope is very steep) and continued sediment deposition will cause sediments to pool. Further deposition on top of uncleared sediment deepens and enlarges the pool and results in some of the 'snow-covered', smothered images seen in this study. If trapped in local minima and with no downhill pathway for shedding, such pools of sediment require substantial wave action to be removed through re-suspension

Sediment re-suspension from the corals' surface was seen in the time series on several occasions where deposits were eroded from the corals' surfaces by waves, allowing an inspection of the fate of the underlying tissues. Previously smothered areas frequently showed bleaching (discolouration), especially in the massive *Porites* spp. and *Turbinaria* spp. Similar bleaching has also been induced by experimental smothering of *Porites* spp. corals and *Montipora aequituberculata* with sediment in aquarium-based experiments [42]. The bleaching *in vitro* was reversible and corals began regaining their colouration over a few weeks if the sediment was removed [42]. Similar re-colouration of bleached tissues, and over a similar time course, was also observed *in situ* during the dredging program. Bleaching in these instances is likely to be caused by either the complete loss of light that occurs under the layers of fine (silt and clay sized) sediments, but not coarse sands, although reduced solute exchange and anoxia cannot be discounted [5, 12].

Different species and morphologies possess different mechanisms to cope with higher levels of sediment deposition. Despite experiencing high levels of smothering and damage some encrusting morphologies were nevertheless quite capable of growing and extending at the distal, leading edges during the dredging program. Similarly, funnel shaped *Turbinaria* spp. often occur naturally with central lesions, referred to as 'sacrificial sediment traps' [46]. This was seen in this study by the formation of the central, basal lesion in the *Turbinaria* spp. which occurred even before the dredging started. Growth and reparation of the central lesion was

observed in such colonies during the dredging phase, when the surface was temporarily sediment-free. Similar patterns of repair were also seen in massive *Porites* spp. during periods when the surfaces were sediment-free in between successive bouts of deposition and smothering.

The production of mucous sheets represents another mechanism to cope with higher levels of sediment in the massive *Porites* spp. These form when sediment loads become too high and subsequent sloughing of the sediment-impregnated mucous sheets effectively resets the surface to a sediment-free one. The response has been likened to the human sneeze reflex [31]. The massive *Porites* spp. are prominent components of reef assemblages in the Indo-Pacific region, and significant frame-work builders[30]. Mucous sheet formation and the ability to episodically reset their surfaces to a clean state is very likely to be one of the reasons why they can live in turbid inshore areas, despite a seemingly poor ability to rid themselves of settling and settled sediments [11]. Mucous sheet formation was also observed in branching *Porites* spp. and use of mucus was frequently seen in some *Turbinaria* spp. However, for the *Turbinaria* spp. the mucus was in the form of mesh like webs or filaments which rolled around the surfaces and collected sediments until shed. Similar use of mucus in *Turbinaria mesenterina* has been seen in laboratory experiments (see Fig 2 in [47]).

The differential effects of sediment smothering on the various morphologies examined indicates a well known trade-off. Corals with angled surfaces are very adept at shifting sediments and preventing smothering, but at the cost of light harvesting. Morphologies with flatter planes of upwards facing surfaces are better suited to light capture but at the cost of greater susceptibility to smothering. Smothering susceptible morphologies appear to have other adaptations to cope, including the ability to grow at the distal edges despite proximal smothering, to tolerate and repair lesions caused by smothering during clement, sediment-free periods, and for some taxa to use mucus and mucous sheet formation to episodically reset their surfaces.

Management implications

Understanding the size of a deposition zone and the area within that deposition where smothering of corals occurs is fundamentally important for defining zones of high impact, for managing dredging using zonation schemes and for impact prediction at the EIA stage. Corals normally keep their surfaces sediment-free, and simple assessments of the percentage of the surface covered by sediment (or mucus) as used in this study is a very practical and rapid survey technique for monitoring and could be conducted by roving diver techniques or even diver-less assessments using remotely operated vehicles. Given the clear association of morality with smothering and the sediment deposition field, the technique could be used to confirm model predictions of sediment transport and fate and the zones of high and medium impact.

Sedimentation is usually cited as one of a suite of different pressures that has caused a decline in many reefs linked to terrestrial runoff and changed land-use practices. However, sedimentation rates have never really been measured on reefs at the appropriate scales [5, 16, 18, 19, 48], but see [22]. Information from sediments traps do not provide information on sedimentation rates in energetic environments, and misinterpretation and misuse of the information could have led to a misunderstanding of coral reef processes [16]. Whilst clearly a key-cause effect pathway for the impact of dredging where sediments are released into low energy water columns where deposition rates will be high, notions of sedimentation stress and damage under terrestrial runoff and natural resuspension events seem incompatible with the sediment-shifting ability shown by many of the corals in this study. This is especially so for the branching morphologies which tolerated extreme levels of sediment deposition. Perhaps for juvenile corals sedimentation poses more of a risk [49, 50], but perhaps also the term

sedimentation is being used too generally to refer to sediment loading and cause-effect pathways associated with turbidity and changes in light quality and quantity, rather than sedimentation *per se*.

Author Contributions

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STANDARD PAPER

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Accounting for environmental uncertainty in the management of dredging impacts using probabilistic dose–response relationships and thresholds

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Abstract

1. Dredging and related activities are common across nearshore marine environments, potentially threatening nearby ecosystems. Regulatory frameworks are essential for minimizing environmental impacts, yet rely heavily on a sound understanding of how ecosystems will respond to environmental stressors, and the thresholds that delineate benign and harmful conditions.
2. Here, we use novel statistical approaches and mathematical tools to account for uncertainty in deriving in situ dose–response relationships and thresholds for the environmental management of dredging, based on estimates of the probability of non-zero mortality of corals during a dredging campaign at Barrow Island, Western Australia. Using modified receiver operating characteristic curves, we derive thresholds with explicit Type I and Type II errors rates, across the full range of primary stress pathways and exposure dimensions (intensity, frequency and duration).
3. Monitoring coral health and mortality can be expensive and water quality indicators are often used to supplement direct receptor monitoring during dredging management. We found strong relationships between coral mortality and a range of water quality exposure metrics, lending support to the use of water quality metrics as management tools for protecting corals during dredging. Metrics based on sediment deposition were more statistically powerful than those based on either light or turbidity, but may be more difficult to implement in practice. Thresholds reflecting aversion to a false sense of security in environmental protection or aversion to the costs of false alarms varied substantially for all exposure metrics examined.
4. *Synthesis and applications*. Strategies for managing environmental harm under uncertainty are critical to achieving an informed risk-weighted balance between environmental protection outcomes and development costs. Our study demonstrates the complexities of how communities respond to variable environmental exposure across a range of pressures in time and space, and the value of integrating probabilistic approaches in environmental management to account for that

complexity and its associated uncertainty. Coral mortality was strongly related to a range of water quality exposure metrics associated with dredging activities, and regulatory thresholds based on water quality can provide a solid and cost-effective foundation for protecting corals during dredging. The probabilistic dose–response relationships and thresholds presented here are the first to be derived from in situ data using dredging related coral mortality and represent a step forward in integrating formal decision science approaches into environmental management.

KEYWORDS

coral mortality, coral reefs, dose–response, dredging, light attenuation, sedimentation, suspended sediments, threshold development, turbidity, water quality

1 | INTRODUCTION

Sediments released into the water column from dredging and dredging-related activities can have a range of effects on nearby marine communities such as corals (Erftemeijer, Riegl, Hoeksema, & Todd, 2012; Jones, Bessell-Browne, Fisher, Klonowski, & Slivkoff, 2016), seagrasses (Erftemeijer & Lewis, 2006) and sponges (Bell et al., 2015). Standard approaches to environmental impact assessment (EIA; Glasson, Therivel, & Chadwick, 2013; Munn, 1979) and management typically use predictive modelling (EPA 2016; Gupta, Gupta, & Patil, 2005), which rely on a firm understanding of how ecosystems will respond to environmental disturbances and clarity in delineating benign and harmful conditions.

Risk management strategies seek to balance the environmental benefits of strict regulation against the costs of alarmism. This is challenging, because ecological systems are inherently complex, and there may be multiple cause–effect pathways by which damage to benthic communities can occur. For dredging, in addition to the direct effects of removal of the seabed, there are indirect effects associated with the generated sediment plumes that impact underlying benthic communities through three highly interconnected pathways: elevated suspended sediment concentrations (SSCs), reductions in benthic light availability through turbidity and higher levels of sedimentation leading to smothering (Jones et al., 2016). Depending on their mode of nutrition, size, shape (morphology) and life-history stage, different benthic taxa may differ in their sensitivity to each of these pathways either alone or in combination.

Water quality conditions during dredging can be spatially and temporally variable (Fisher, Stark, Ridd, & Jones, 2015; Jones, Fisher, Stark, & Ridd, 2015). Environmental effects may manifest as a result of the overall intensity (*I*), as well as the frequency (*F*) and the duration (*D*) of a disturbance, and it is important to quantify environmental pressure across all three IDF dimensions. An IFD approach is used in defining rainfall events (Koutsoyiannis, Kozonis, & Manetas., 1998) and examining climate change impacts (Webster, Holland, Curry, & Chang, 2005) and has also been attempted in the development of guidelines for dredge material disposal (McArthur, Ferry, & Proni, 2002). Given the transient, ephemeral nature of dredging related turbidity events (Jones et al., 2015), exposure of each of the three cause–effect pathways need to be considered for each of the three IFD dimensions.

For dredging, thresholds are critical in EIAs for delineating spatial boundaries of allowable impact (EPA 2016) and for adaptive operational management as triggers of response actions (such as stopping, relocation or modifying the mode of operation of dredging). Often thresholds are set without explicit consideration of the uncertainty in the biological responses or statistical power. Mathematical tools such as receiver operating characteristic (ROC) curves enable thresholds to be derived under probabilistic uncertainty (Swets, Dawes, & Monahan, 2000). These techniques have been applied to diagnostic tests in a medical context, but despite similar theoretical considerations, there are few examples of their use in an environmental (regulatory) setting (but see Connors & Cooper, 2014). In general, those who are more averse to the costs of environmental damage arising from a false sense of security are inclined to advocate for thresholds with a high true-positive rate. Conversely, those who are averse to the financial costs (i.e. stopping dredging) associated with a false alarm are inclined to thresholds with a low probability of false positives. In the pervasive circumstance of an imperfect metric, ROC curves can be used to make informed judgments on how to balance these considerations in a risk-weighted framework (Burgman, 2005).

In this analysis, we use novel statistical approaches and mathematical tools to derive in situ dose–response relationships and critical thresholds for the management of dredging using information collected during a capital dredging project that occurred near the reefs around Barrow Island (Western Australia). The monitoring associated with this project went beyond that required by the regulatory framework and involved continuous water quality monitoring concurrent with health measurements of hundreds of corals examined at two weekly intervals across 26 sites (Styan & Hanley, 2013) located along a gradient from 200 m to up to 30 km away from the dredging.

We examine dose–response relationships and thresholds across a suite of environmental exposure metrics representing the three primary cause–effect pathways (turbidity as a proxy for SSCs, light

attenuation and sediment deposition) and three exposure dimensions (intensity, frequency and duration), and we test their in situ predictive ability. We base the dose–response relationships on the probability that any given site showed non-zero mortality and build a simulation procedure to derive thresholds using ROC curves with explicit probabilities of Type I and Type II errors.

2 | MATERIALS AND METHODS

Dredging was undertaken at Barrow Island over a 530-day period from 19 May 2010 to 31 October 2011, with water quality and coral health data collected at 26 sites throughout the duration, and for extensive baseline periods (Figure 1). Water quality data included measurements of turbidity as nephelometric turbidity units (NTU), photosynthetically active radiation and an optical backscatter sediment deposition sensor (Ridd et al., 2001) (Appendix S1). Dredging impacts on water quality were spatially asymmetric, with plumes largely undergoing a unidirectional flow to the south. Impacts on water quality were evident up to 19.6 km S and 2.1 km N of the dredging activity (Fisher et al., 2015). All sites outside these reported bounds were, therefore, considered non-impacted control or reference sites (white symbols, Figure 1), providing they were also more than 5 km away from the dredge material placement site (seven sites). Within the areas influenced by the plumes, we focus on those sites nearest to the dredging activity (all MOF and LNG sites) and those making up a transect to the south (TR, DUG and BAT), because these combined represent the strongest gradient in water quality impacts (Fisher et al., 2015; black symbols, Figure 1, 15 sites).

Our analytical approach to develop dose–response relationships and management thresholds involved first deriving water quality and coral mortality metrics for each site, combining these as dose–response relationships and evaluating these against model selection methods used to determine which water quality metrics were the best predictors of coral mortality (Figure 2). The best metrics were analysed using ROC curves to derive thresholds with explicit probabilities of Type I and Type II errors (Figure 2).

2.1 | Coral mortality metric

Live coral cover data were obtained from images of individual corals, taken approximately every 14 days throughout the dredging phase (Figure 3) yielding an average 37 repeated images throughout the monitoring project (range 7–51). These images were scored as a percentage of live coral tissue and subsequently smoothed to remove anomalies (Figure 3 and Appendix S2). Following data cleaning (Appendix S2), the number of coral heads monitored at individual sites around Barrow Island ranged from 18 up to 66 (Table S2.1). Coral heads were divided into 15 family-morphology groups (i.e. branching or massive growth forms) of which four had sufficient replicate colonies (>3 colonies) across enough control sites (>2 control sites) for analysis (branching Acroporidae, 251 colonies; branching Pocilloporidae, 58; massive

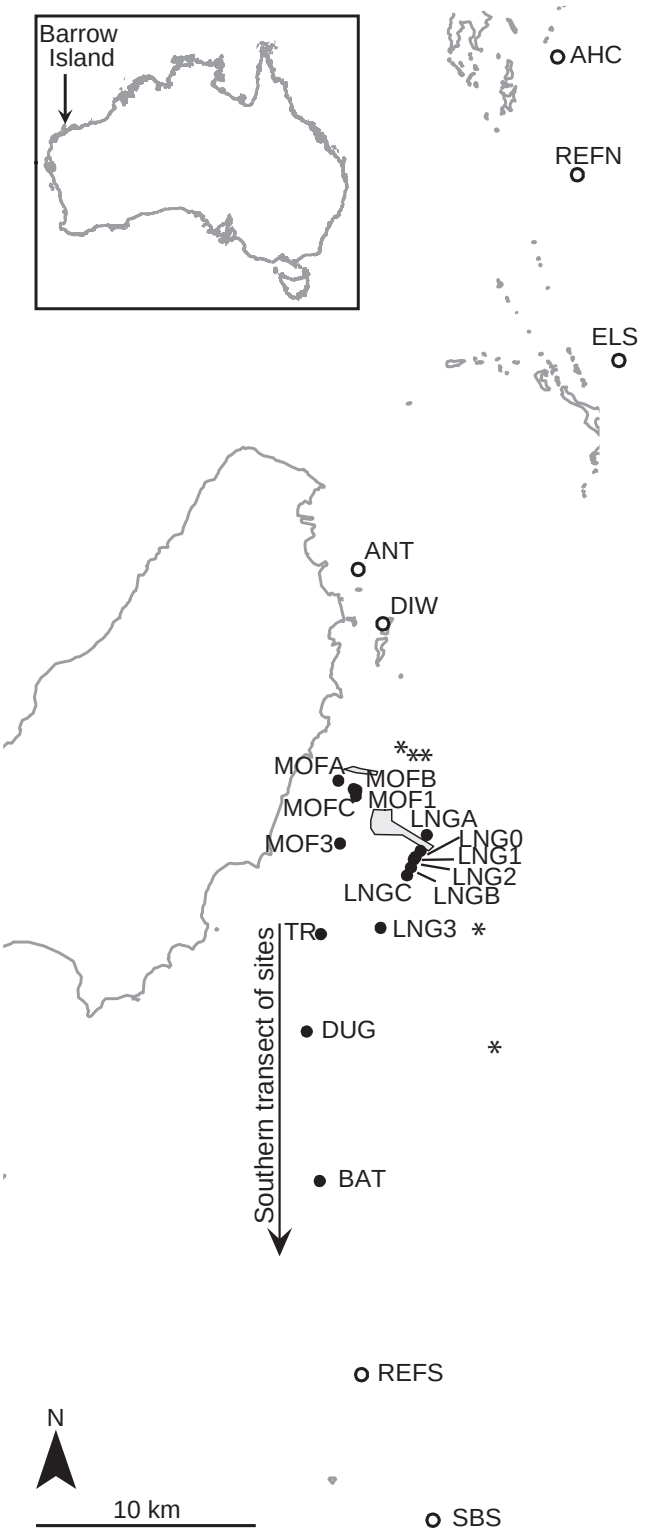

FIGURE 1 Water quality and coral health monitoring sites at Barrow Island. Control sites (non-impact, open circles, *N* = 7) included all those >2.1 km N and >19.6 km S of the dredging activity (Fisher et al., 2015). Potential impact sites (closed circles, *N* = 15) included all those in the immediate vicinity and south of the dredging. Sites close to dredging on the Northern side and/or too close to the spoil disposal site were not included in the analysis (shown as *)

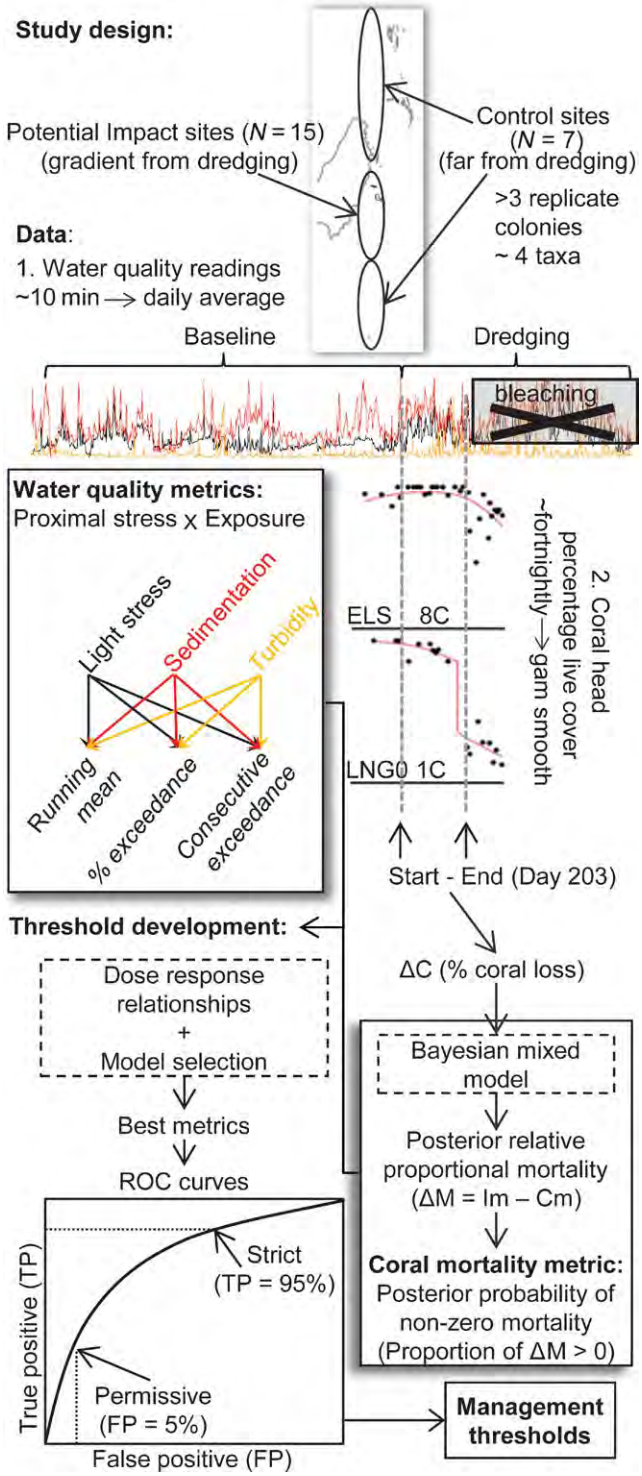

FIGURE 2 Flow diagram detailing the analytical steps used to derive dose–response relationships and management thresholds. [Colour figure can be viewed at wileyonlinelibrary.com]

Faviidae, 134; and massive Poritidae, 510 colonies; Appendix S2). A moderate coral bleaching event occurred during dredging in the summer of 2010–2011, due to a warm water anomaly (Appendix S3). To avoid confounding thermally induced bleaching with dredging effects, we focused our analyses on the first 203 days of dredging before any bleaching was observed. While only representing a portion of the total dredging period, these first 203 days show clear levels of elevated turbidity associated with dredging (Figure 3; Jones et al., 2015). In order to examine how water quality conditions during dredging relate to coral mortality, it was necessary to factor out natural mortality that occurs even in the absence of dredging-related pressures and calculate the likelihood that there was in fact elevated mortality at any of the potential impact sites. This was achieved through a series of steps (outlined in Figure 2 and explained in detail in Appendix S2). First, the percentage of live coral loss (ΔC) was calculated as the difference in percentage live cover of each coral head at the start of dredging compared to day 203 (prior to the start of the bleaching event) using the smoothed live coral trajectories. We assumed that the percentage live coral loss (ΔC) was a binomial random variable, dependent on percentage cover at the start of dredging (T_i) and probability P_i (Equation 1). This was modelled via a logit link as a function of the fixed factor CvI (distinguishing between the control locations and the specific potential impact location being examined) and a random site level offset a_{ji} to account for the non-independence of coral heads collected at individual control sites (Equation 2). Parameters β_0 and β_1 are the estimated parameters of the logistic regression (Equation 2).

$$\Delta C_{ij} \sim \text{binomial}(T_{ij}, P_{ij}) \quad (1)$$

$$\text{logit}(P_{ij}) \sim \beta_0 + \beta_1 \times \text{CvI}_{ij} + a_{ji} \quad (2)$$

Random normal diffuse non-informative priors were used for the parameter estimates for β_0 and β_1, with half-Cauchy priors for the random site variance σ^2. To aid convergence, starting values of β_0 and β_1 were based on the equivalent mixed model fit using glmer from the lme4 library (Bates, 2010) with starting values for the random control site level deviations taken from a normal distribution with mean of zero and standard deviation of 0.001. We ran five MCMC chains each with 100,000 iterations and 30,000 iterations were discarded as "burn-in." Chain mixing was assessed visually using trace plots and found to be acceptable. Relative proportional mortality (ΔM) for each sites was then calculated from the MCMC iterations as the inverse logit of the difference in the model estimated proportional coral loss at the impact location being examined (I_m) minus the modelled coral loss at the control locations (C_m).

$$\Delta M = I_m - C_m \quad (3)$$

Positive values of ΔM (>0) indicate that mortality at the site of interest was in excess of that observed at control locations, and negative values (<0) indicate that mortality at a site was less than that observed across the control locations. Values near zero indicate that mortality at the impact and control locations are similar. Here, we assume a hypothetical regulatory setting where the local environment authority has an interest in any non-zero adverse impact. In line with this regulatory setting, the posterior probability of non-zero mortality is calculated as the proportion of the MCMC sample of relative proportional mortality (ΔM) that is above zero. The approach can be more broadly applied to any setting by calculating the proportion of the MCMC sample above any magnitude of impact that may be of interest (such

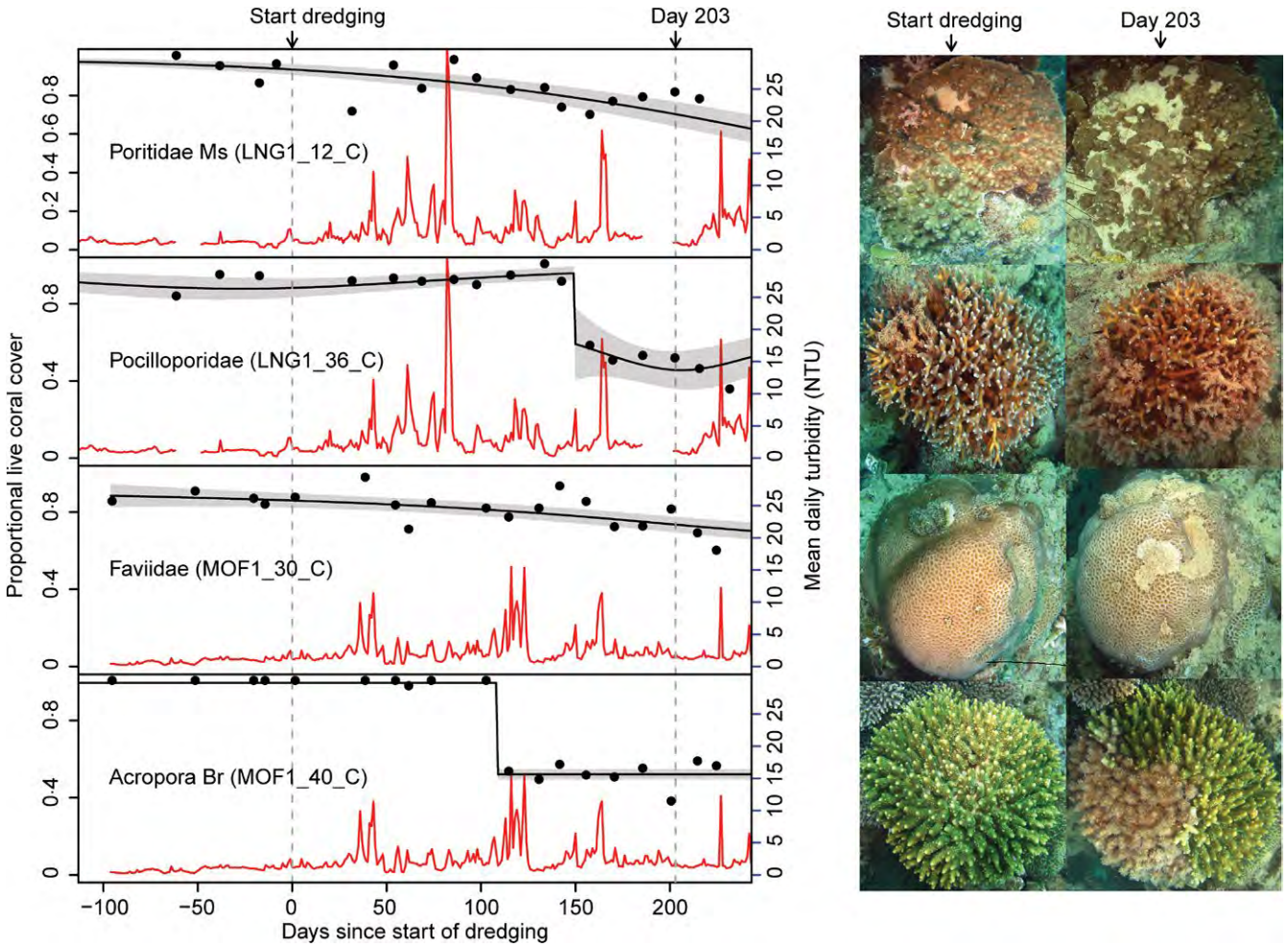

FIGURE 3 Example coral mortality and water quality data used for analyses, showing time series of proportional live coral cover measurements for representative colonies of each taxa (black circles) and mean daily turbidity (NTU; red lines). Solid black lines represent GAM smooths (see Appendix S2 for details) with grey bands indicating 95% confidence bounds, fitted using a beta distribution to the proportion of live coral cover. Right hand panels show images of each colony at the start of dredging and immediately preceding the start of bleaching (day 203, used as the end point in the present analysis, Figure 2)

as a relative decline of 30% partial mortality). Posterior probabilities of non-zero mortality were calculated individually for each potential impact site, for each family/morphology group for which there were sufficient replication (greater than three colonies at the impact site and more than two control locations). Consistent with our assumed regulatory setting (where the authority is interested in any non-zero impact), we used the maximum observed mortality values for any one taxonomic group to derive dose–response relationships and thresholds, as it was deemed that this would have the greatest chance of yielding thresholds protective of the community as a whole.

2.2 | Water quality exposure metrics

A set of candidate water quality exposure metrics were derived from the time series of water quality readings, yielding three proximal stress indicators capturing the various dredging related mortality pathways for corals, including benthic light, (Daily light integral — DLI, mol quanta m^{-2} day^{-1}) expressed as a stress index between 0 (corresponding to a maximum possible subsurface DLI of ~30 mol quanta m^{-2} day^{-1}) and 1 (complete loss of all benthic light), turbidity [as a surrogate for SSC, based on log (Nephelometric Turbidity Units + 1)] and sediment deposition (daily integrals of the square-root of sediment surface density — SSSD, mg cm^{-2}, Ridd et al., 2001) from the optical backscatter sensors (Appendix S1).

Values for each proximal stress indicator were available daily at each site and required aggregation to a single site level value for subsequent analyses. Here, we explore a wide range of aggregation strategies, focussing on three-dimensions of exposure: intensity, frequency and duration. Intensity of exposure was captured as the worst case value of each proximal stress metric experienced at a site. Across each daily time series at a site, worst case values were determined from the raw daily mean as well as from daily running means calculated using increasing time spans (1, 7, 14, 21, 30 and 60 days, Appendix S1), yielding intensity metrics capturing a range of temporal scales of exposure. Frequency of exposure was captured as percentage exceedance of baseline conditions, with exceedances calculated across a range of

percentiles (P): P_{80}, P_{90} and P_{95}, representing increasing extremes of these baseline conditions. Baseline conditions for defining exceedance were calculated as global values, with a single value used across all sites (Table S1.2), because baseline time series were relatively short at some sites. Duration was captured as worst case consecutive days of exceedance of baseline conditions, across a range of percentiles (P): P_{80}, P_{90} and P_{95} (Appendix S1). This yielded 33 possible exposure metrics that captured dredging related pressure for each site.

2.3 | Threshold development

All 33 exposure metrics were examined as predictors of the posterior probability of non-zero coral mortality (calculated from the Bayesian mixed model analysis, see 2.1 above) in individual dose–response relationship, fitted at the site level (N = 22; 15 potential impact + 7 control sites, Figure 1) using a two parameter logistic regression:

$$y_i = 1/(1+\exp(-(-a+b\times x_i)) \quad (4)$$

where y_i is the posterior probability of non-zero mortality at site i, and x_i is the value at site i for the exposure metric being examined.

The predictive utility of all metrics was compared using AICc (Burnham & Anderson, 2002) and R^2 values. Water quality metrics showing the strongest predictive relationship with coral mortality for each proximal stressor (light, turbidity and deposition) and each exposure dimension (running mean, percentage exceedance and consecutive exceedance) were selected for formal threshold development. A quantitative decision science approach was used for threshold development, where ROC curves (Swets et al., 2000) were calculated and thresholds derived based on scenarios for true-positive (1 — Type II error) and false-positive probabilities (Type I error; Appendix S4). Existing software for calculating ROC curves is based on a binary response variable (i.e. impacted or not-impacted) without capturing any measure of uncertainty in the assignment. We developed a simulation procedure where ROC curves were calculated by randomly allocating sites to either impact or non-impact categories, based on their observed posterior probability of non-zero mortality (Appendix S4). Calculated ROC curves, along with associated statistics and derived thresholds, were represented by 95% probability distributions reflecting uncertainty in impact assignment.

Ideally, ROC curves should be coupled with a cost model that captures the relative consequences of false negatives (i.e. failing to detect an impact where one exists; Type II error) and false alarms (Type I error). A detailed cost analysis is beyond the scope of this study. We illustrate key concepts by exploring two contrasting regulatory perspectives: a "strict" threshold (true-positive probability = 0.95) and a "permissive" threshold (false-positive probability = 0.05). These thresholds are hypothetical (having no basis in the actual costs of Type I and Type II errors); however, conceptually, the strict threshold will be relevant where the avoidance of adverse impacts on the environment is placed at a premium, such that the costs of missing a potential impact far outweigh the costs of false alarms. The permissive threshold represents a situation where false alarms are very costly. We emphasize that water quality attributes are powerful, but imperfect predictors of benthic coral health. For any imperfect predictor, a threshold set at a level that avoids false alarms (Type I errors) inevitably carries elevated risks of failing to detect actual impacts and vice versa. Ultimately, managers and stakeholders need to reconcile the costs of Type I and Type II errors and the costs of data acquisition.

3 | RESULTS

3.1 | Coral mortality and exposure metric characteristics

The posterior probability of non-zero mortality across potential impact sites varied widely (Appendix S2, Figure S2.2), but was only greater than 0.5 at sites very near the dredging (<2 km, Figure 4). Only one site had >0.95 probability of non-zero mortality (MOFA, 0.56 km), although 22% of sites had probabilities of mortality >0.75 (Figure 4). The sites furthest from dredging for which there was more than a 0.50 probability of non-zero mortality were LNGC (1.4 km, massive Poritidae) and MOF3 (1.46 km, branching Acroporidae, Figure 4). At any given site, there were differences in the posterior probability of mortality among taxa, with few patterns emerging (Figure 4). Mortality was highest for the massive Poritidae and the Pocilloporidae, for which 43% and 40% of sites has >0.50 probability of non-zero mortality, compared to only 22% and 27% of sites for the branching Acroporidae and the Faviidae/Mussidae groups, respectively (Figure 4).

There was a relatively strong gradient of water quality conditions with distance to dredging (Figure 4). Sites formed two primary clusters, with all control sites (labels shown in green) along with the site BAT forming a group of sites with very low values across all metrics (Figure 4). There was considerable structure in the remaining cluster, with sites forming three groups of varying levels of water quality exposure (Figure 4).

3.2 | Dose–response relationships and thresholds

All 33 water quality exposure metrics were relatively strong predictors of the posterior probability of non-zero coral mortality, with all metrics scoring at least 300 AICc units lower than a null model (Appendix S5). For each of the three proximal stressors (deposition, turbidity and light), the best metric for each exposure dimension was deposition: 60-day running mean, percentage exceedance of P_{80} (% of days >0.04 SSSD) and consecutive days where deposition exceeded P_{95} (maximum consecutive days >0.19 SSSD); turbidity: 14-day running mean, percentage exceedance of P_{80} (% of days >1.8 NTU) and consecutive days where turbidity exceeded P_{95} (maximum consecutive days >4.8 NTU); and light: 14-day running mean light, percentage exceedance of P_{80} (0.58 light, <2.2 DLI) and consecutive days where light stress exceeded P_{80} (maximum consecutive days >0.79 light stress, <0.3 DLI). These nine exposure metrics were used in ROC curve construction and threshold development (Appendix S5 contains all 33 dose–response model fits), with all nine showing high accuracy (area under the curve, median values >0.8 in all instances), although 95% confidence bands indicated considerable variability, with accuracy varying by 3%–7% (coefficients of variation across all ROC curves constructed; Figure 5). Accuracy was highest for the three deposition-based metrics, with

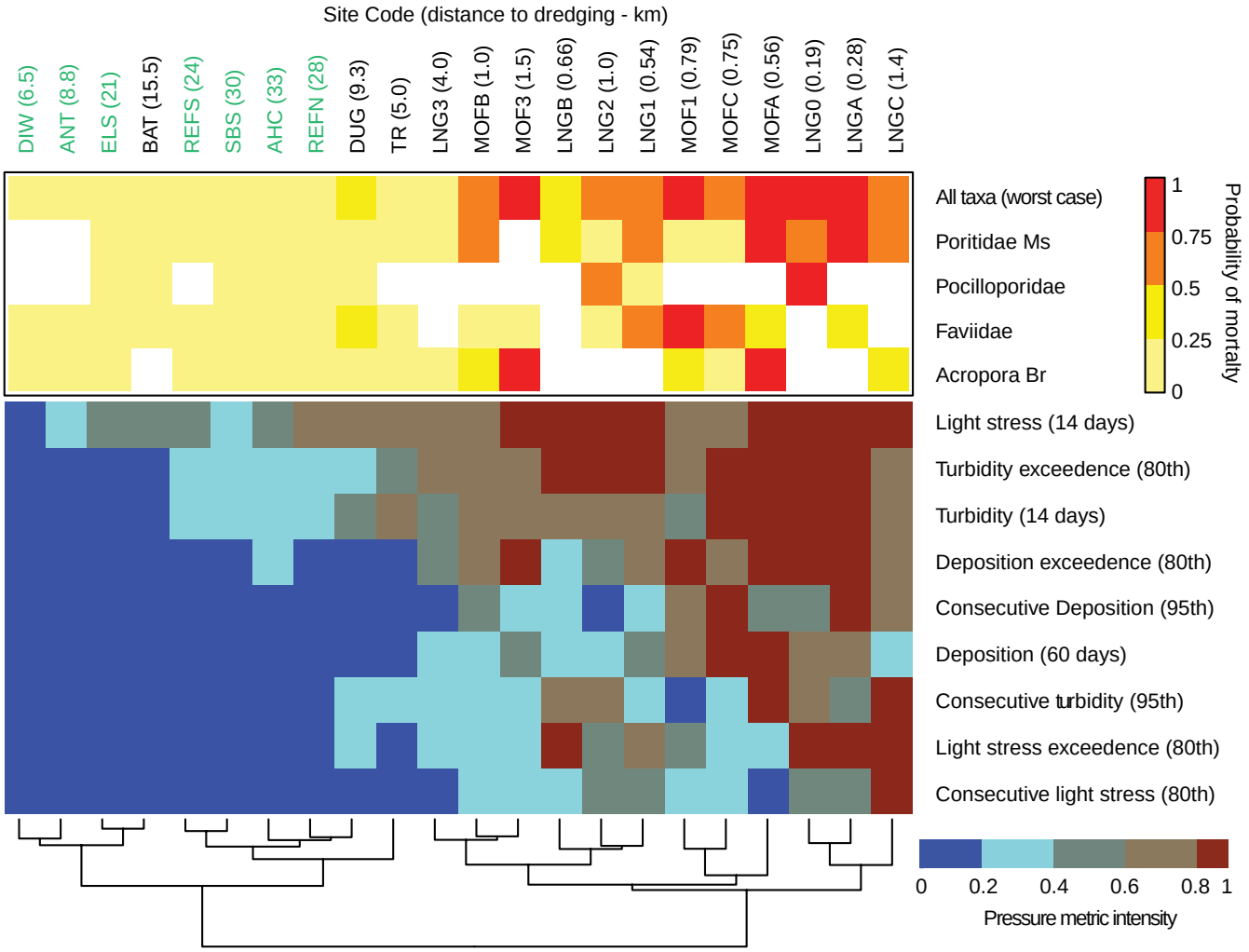

FIGURE 4 Heat maps summarizing the posterior probability of non-zero coral mortality (upper) and nine water quality exposure metrics (lower) across sites. White boxes indicate sites with insufficient data to assess mortality for that taxa. Control sites labelled in green and potential impact sites black. Dendrogram shows a Euclidian hierarchical cluster analysis (complete linkage), with sites ordered from the lowest to the highest branch means within each cluster

exceedance the strongest, followed by 60-day running mean and consecutive deposition (Figure 5), reflecting the relative strengths of the dose–response relationships (R^2 values, Figure 5). High accuracy for deposition-based metrics is a result of their relatively strong distinction between the impact and non-impact sites (Figure 6a, Appendix S6).

Estimated strict (true-positive probability = 0.95) and permissive (false-positive probability = 0.05) thresholds differed across all metrics examined (Table 1). With the exception of the deposition-based metrics, 95% confidence bands of the estimated strict and permissive thresholds did not overlap (Table 1). Strict thresholds for the running mean stress metrics were 0.31 for deposition (60 days), 0.8 for turbidity (14 days) and 0.66 for 14-day light, and are equivalent to 0.1 SSSD, 5.3 NTU and 1.2 DLI (Table 1). Permissive thresholds for the running mean stress metrics were substantially higher, at 0.63 for deposition (60 days), 1.05 for turbidity (14 days) and 0.77 for light (14 days) and equivalent to 0.4 SSSD, 10.2 NTU and 0.37 DLI (Table 1). Strict thresholds based on the percentage exceedance of baseline conditions varied from 25% of days (deposition >0.04 SSSD) to 45% of days (turbidity >1.8 NTU), whereas permissive thresholds reached as high as 64% of days (turbidity >1.8 NTU, Table 1). Thresholds based on the number of consecutive days of exceedance of baseline conditions were quite similar for deposition (strict: 2.5 days; permissive: 8.2 days) and turbidity (strict: 2.9 days, permissive: 10.3 days) but were much higher for light (strict: 9.9 days; permissive: 34 days), which was based on an 80th rather than a 95% percentile (Table 1).

When a strict threshold is set at a true-positive probability of 95%, corresponding false-positive probabilities (false alarm rates) ranged from 11% to 50% (Table 1). Conversely, when a permissive threshold is set at a false-positive probability of 5%, corresponding true-positive probabilities (statistical power) ranged from 12% to 49% (Table 1).

4 | DISCUSSION

Coral mortality was strongly related to a range of exposure metrics capturing changes in water quality associated with dredging activities,

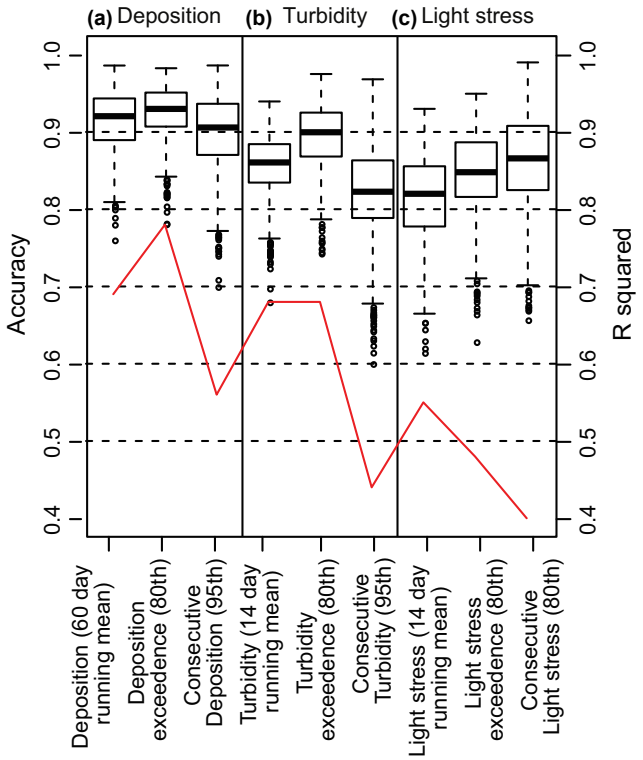

FIGURE 5 Accuracy (area under the curve) of ROC curves (boxplots) and R^2 values for dose–response relationships (red line) for nine water quality exposure metrics. [Colour figure can be viewed at wileyonlinelibrary.com]

with metrics explaining nearly 80% of the variability in the posterior probability of non-zero coral mortality among the sites examined. ROCs curves were correspondingly powerful, with accuracy values ranging from 0.81 to 0.94. It is clear that regulatory thresholds based on water quality can provide a solid and cost-effective foundation for protecting corals during dredging.

Here, we have explored exposure metrics based on the three primary dredging-related cause–effect pathways of impact on corals: sediment deposition, turbidity (as a proxy for SSCs) and reductions in benthic light. Of these, deposition-based metrics were the strongest predictors and had ROC curves with the greatest accuracy. Low light impacts corals through reduced photosynthesis of their endosymbiotic dinoflagellates (providing the coral host with up to 90% of its energy, Muscatine, 1990) and elevated SSC can cause polyp retraction, reducing heterotrophic feeding (Anthony, 2000; Mills & Sebens, 1997). However, sediment deposition and subsequent smothering of corals causes similar effects, along with additional effects associated with tissue anoxia and changes in coral tissue pH (Weber et al., 2012), that can lead to lesion formation and partial mortality. Despite the predictive strength of deposition-based metrics, sediment deposition is inherently difficult to measure in situ (Ridd et al., 2001). While the sensory logger-based SSSD measurements used here provide information on sediment deposition over appropriate time frames (days), recorded SSSD values are dependent on the physical properties of the sensor surface and provide relative rather than absolute values. There is a pressing need to develop instrumentation to measure sediment deposition rates with appropriate sensitivity and adequate temporal resolution. In the absence of reliable deposition measurements, for now we must rely on light and turbidity, despite their lower statistical power. Both turbidity and light are relatively easy to measure in situ using well-established, highly repeatable technology.

Our analyses are based on the worst case observed mortality across any one of the four taxonomic groups examined at a given site. The three cause–effect pathways may operate to lesser or greater extents on corals of different growth forms and/or other life-history characteristics. For example, branching corals can quite easily shift sediments from their surfaces and corals with larger polyps may also have greater capacity to shift sediment, making them more resilient to sediment stress (Stafford-Smith & Ormond, 1992). However, corals more reliant on photosynthesis than heterotrophic nutrition (Chappell, 1980) may be more susceptible to reduced light associated with dredging. Thus, variation in mortality of taxa among sites is likely due to differences in the balance between sedimentation, turbidity and light reduction. Were taxa-specific analyses possible, it seems likely that different proximal stressors would be more relevant to some coral types than others. However, given all of the exposure metrics examined are highly interrelated, it is not surprising that they all show similarly high accuracy for deriving thresholds, regardless of the specific cause–effect pathway they represent.

We have explored a wide range of exposure metrics representing different levels of intensity, frequency and duration. The metrics we defined conceptually overlap in this context: running means across different time frames capture elements of both intensity and duration; and different baseline percentiles to define the frequency and duration of exceedances capture elements of intensity. Across the range of values explored our metrics capture (to varying degrees) elements corresponding to chronic (longer running means and lower percentile values) as well as acute stress (shorter running means and higher percentile values). Relationships were strongest for a 60-day running mean deposition, and 14-day running means were strongest for turbidity and light stress. In terms of percentage exceedance-based exposure metrics, the 80th percentile values provided the best results, resulting in thresholds representing high exceedance frequencies (15%–70% of days) of relatively low baseline values. Both the longer running mean time scales and a P_{80} exceedance frequency imply that chronic, rather than acute, events may be important in this system. However, the best estimated threshold values for consecutive exceedance for deposition and turbidity were based on P_{95}, yielding thresholds of consecutive days of exposure in the order of 2–10 days which represent acute impacts. Chronic stress metrics that average over long time frames (e.g. 60-day running mean deposition) may be very useful in pre-dredging modelling studies aimed at predicting spatial patterns in mortality for the purposes of environmental approvals and delineating zones of expected impact (EPA 2016). From an operational perspective, where thresholds are used to trigger response actions during dredging, long-term thresholds may have limited utility. By the time a 60-day threshold has been exceeded, there may be few alternative actions left available to remedy the situation, implying that the "best" depends on both the statistical performance, the intended operational

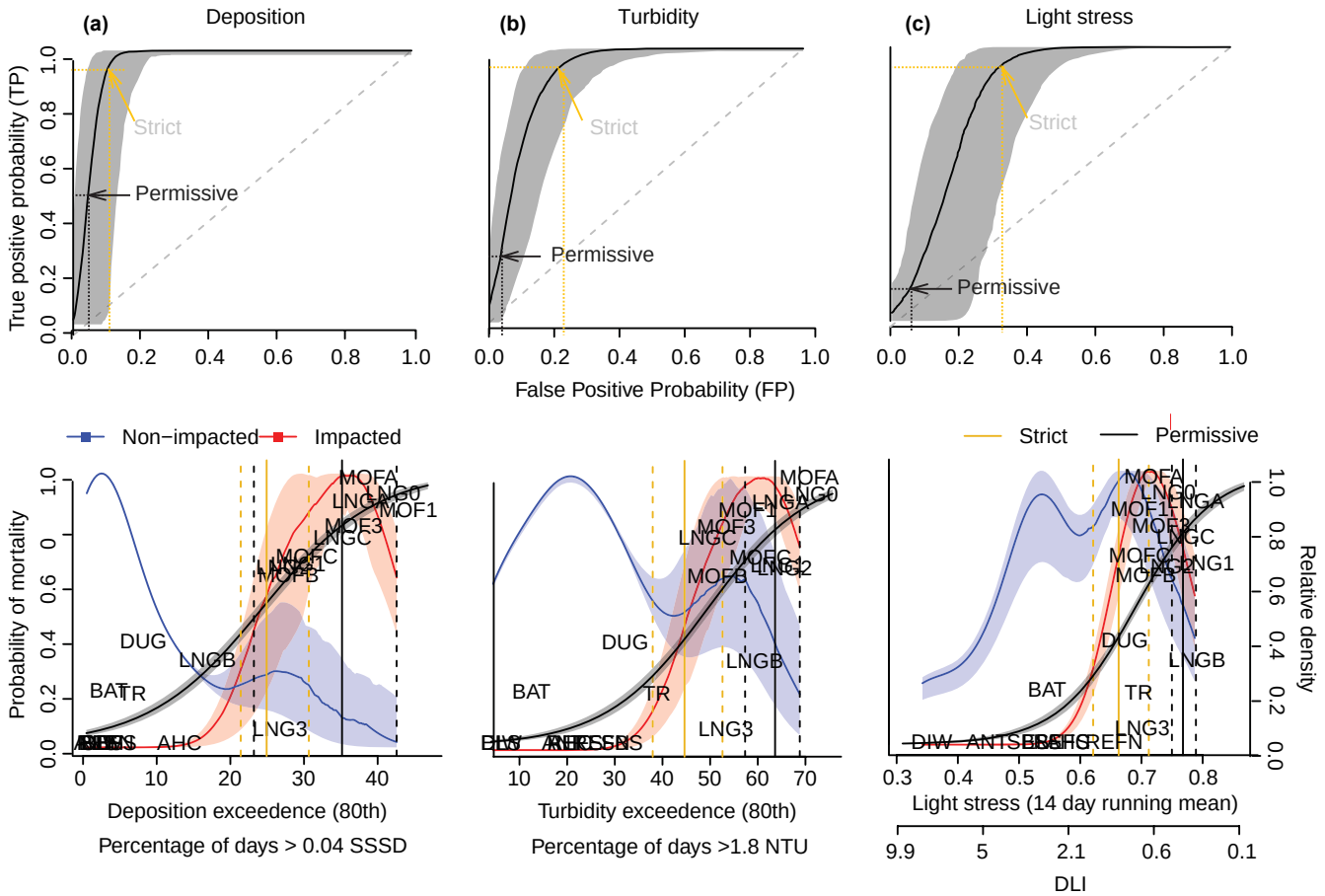

FIGURE 6 ROC curves (upper panels), dose–response relationships (lower panels, black curves with grey 95% CI), posterior probability densities for non-impact (blue curves with 95% CI) and impact (red curves with 95% CI) locations, and derived strict (TP probability = 95%, yellow vertical lines) and permissive (FP = 5%, black vertical lines) thresholds (dashed vertical lines indicate 95% CI) for representative water quality exposure metrics for each of three proximal stress indices (deposition, turbidity and light stress)

use of the threshold and the decision context. As such, short-term metrics (such as 14-day running means) may be more useful as operational management triggers.

For practical reasons, the exceedance metrics used here were calculated using percentile values of the global baseline dataset across all sites (baseline time series were short at some sites). Exceedances could be calculated using baseline values of individual sites or relative to "controls" in real time. Thresholds based on relative values assume communities are suitably adapted to local conditions and have the capacity to withstand similar levels of deviation in exposure. Setting thresholds based on this assumption has inherent danger as corals growing in marginal environments (i.e. turbid reef zones) may be closer to their physiological limits. Barrow Island is a relatively clear water environment, even compared to nearby reefs at Cape Lambert and the Dampier Archipelago (Fisher et al., 2015), and it remains to be seen if corals persisting in naturally turbid waters (Albert, Fisher, Gibbes, & Grinham, 2015; Browne, 2012) are more or less susceptible to sediment-related stressors, and whether the thresholds derived here may be transferable to other systems. Setting thresholds relative to control locations also means the synergistic effects of natural disturbance regimes, such as turbidity generated during cyclones and/or extreme rainfall will not be effectively managed. Natural disturbance regimes are an inherent feature of coral reef ecosystems, but can be altered through human activities, transforming pulse events into persistent or even chronic disturbances (Nyström, Folke, & Moberg, 2000). Managing local impacts in the face of multiple natural, human induced and altered stressors across broad spatial and temporal scales (Hughes & Connell, 1999) remains a considerable challenge in environmental science.

The analysis here is based on probabilities of non-zero partial mortality (relative to mortality observed at control sites), and the derived thresholds are, therefore, applicable when the goal is to prevent any elevated mortality associated with dredging, as would be the case where regulations stipulate that water quality may be impacted by dredging but should not result in coral mortality (i.e. for an identified "zone of influence" EPA 2016). Our approach could be easily modified to establish thresholds based on probabilities of mortality effects greater than zero (e.g. a 30% relative decline), where this is required under a different regulatory setting. Actual mortality estimates were relatively low during the first 203 days of dredging activity, with only a single site having greater than 0.95 probability of non-zero mortality, with median relative partial mortality at this site still less than 20% relative to controls (Figure S2.1). Basing thresholds on the probability

TABLE 1 Derived threshold values for nine water quality exposure metrics, based on the posterior probability of non-zero coral mortality. True-positive (TP), false-positive (FP) percentages and estimated threshold values are shown for both strict and permissive thresholds, with units showing threshold values in the original units of the variable underlying each exposure metric

Metric	Strict threshold (TP = 95%)			Permissive threshold (FP = 5%)		
	FP (%)	Value	Units	TP (%)	Value	Units
Deposition (60-day run mean)	21 (7–44)	0.31 (0.23–0.45)	0.1 SSSD (0.05–0.22)	49 (14–91)	0.63 (0.36–0.85)	0.4 SSSD (0.13–0.71)
Deposition exceedance (P_{80})	11 (5–22)	25 (21–31)	% days >0.04 SSSD	49 (1–95)	35 (23–43)	% days> 0.04 SSSD
Consecutive deposition (P_{95})	22 (4–95)	2.5 (0.6–4.6)	Consecutive days >0.19 SSSD	37 (0–98)	8.2 (2.9–12)	Consecutive days >0.19 SSSD
Turbidity (14-day run mean)	32 (21–49)	0.8 (0.7–0.9)	5.31 NTU 4.2–6.4	24 (2–45)	1.05 (1.01–1.1)	10.2 NTU (9.2–11.8)
Turbidity exceedance (P_{80})	24 (13–41)	45 (38–53)	% days >1.8 NTU	32 (4–65)	64 (57–69)	% days >1.8 NTU
Consecutive turbidity (P_{95})	50 (31–73)	2.9 (1.5–4)	Consecutive days >4.8 NTU	34 (1–63)	10.3 (7.4–14.7)	Consecutive days >4.8 NTU
Light (14-day run mean)	33 (21–52)	0.66 (0.62–0.71)	1.15 DLI (1.63–0.72)	12 (0–33)	0.77 (0.75–0.79)	0.37 DLI (0.47–0.28)
Light exceedance (P_{80})	35 (21–52)	17 (12–25)	% days >0.58 (<2.2 DLI)	25 (0–62)	59 (42–73)	% days >0.58 (<2.2 DLI)
Consecutive light (P_{80})	31 (15–49)	9.9 (6.2–15.3)	Consecutive days >0.58 (<2.2 DLI)	23 (0–76)	34 (18–66)	Consecutive days >0.58 (<2.2 DLI)

of a target mortality provides an effective means of accounting for environmental uncertainty that would be difficult to achieve through conventional methods based on observed mortality. The approach could also be applied to other sublethal indices of coral health, such as mucus production (Bessell-Browne, Negri, et al., 2017) and bleaching (Bessell-Browne, Fisher, Duckworth, & Jones, 2017) or aggregate indices of coral condition (Ferrigno et al., 2016).

Strong relationships between water quality exposure metrics and the probability of coral mortality do not imply error-free thresholds for discriminating between benign and harmful conditions. We encourage regulators to embrace probabilistic uncertainty in risk-weighted management frameworks. Here, we have shown how modern Bayesian statistical methods can be combined with formal mathematical approaches in decision science to derive management thresholds that can be explicitly interpreted in terms of their Type I and Type II errors.

The probabilistic thresholds derived here are the first to be derived from in situ data using measured dredging related coral mortality and represent a step forward in terms of integrating formal decision science approaches into environmental management. However, a range of challenges remain with implementing this approach more broadly. First, our approach is highly dependent on partitioning out natural mortality using appropriate control sites, which may not always be available or appropriately identified. Second, the collection of monitoring data on the scale undertaken at Barrow Island is rarely carried out in practice and it is unlikely that similar analyses will be possible at other locations. The thresholds derived here are likely specific to the mortality and exposure profiles that occurred at Barrow Island and studies exploring variation in the sensitivity of individual taxa and communities to dredging related pressures would provide insight into the extent to which the thresholds derived here may apply more generally, or if location-specific thresholds are required. Our turbidity threshold of 45% of days exceeding 1.8 NTU (P_{80}, SSC = ~3 mg L^{-1}) is reasonably coherent with the long-term initial water quality criteria used at Barrow Island (20 days in 60 ≥ 5 mg L^{-1}) (Chevron 2009). Furthermore, the running 14-day mean DLI threshold of 1.15 (mol photons m^{-2} day^{-1}) is also consistent with recent experiments on corals from the Great Barrier Reef showing partial mortality and bleaching responses after 28 days of exposure to light levels below 1.1 DLI (Bessell-Browne, Fisher, et al., 2017), suggesting that the thresholds we have derived may have broader applicability.

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AUTHORS' CONTRIBUTIONS

All authors conceived the ideas and designed elements of the methodology; R.F. and P.B.B. analysed the data; R.F. led the writing of the

manuscript. All authors contributed critically to drafts and gave final approval for publication.

DATA ACCESSIBILITY

Raw data used in these analyses are stored at the PAWSEY data centre (https://data.pawsey.org.au) with access granted to the authors under a data sharing agreement through the Dredging Science Node of the Western Australian Marine Science Institution. Requests to access raw data can be made at https://www.wamsi.org.au/data-and-information. Metadata are available through the Australian Institute of Marine Science Data Centre at http://data.aims.gov.au/metadataviewer/uuid/08358639-ddfa-4746-9861-dc8e6f86714f.

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SUPPORTING INFORMATION

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

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Appendix S1. Detailed methods - Water quality exposure metrics.

Water quality data collected at Barrow Island included turbidity (Nephelometric Turbidity Units, NTU) and light (photosynethically active radiation, PAR), and were processed following (Jones et al. 2015a). Data was also available on deposition rates as measured as settled sediment surface density (mg cm^{-2}) calculated from an optical backscatter sensor (Ridd et al. 2001) and aggregated to a daily integral (mg $cm^{-2}d^{-1}$).

Elevated suspended sediment concentrations associated with dredging activities can impact corals through three cause-effect pathways associated with turbidity generation activities (Jones et al 2015a). These include impacts on light availability (LA), suspended sediments (SS), and sediment covering (SC). Here were derived three proximal stress indices that represent all three cause-effect pathways.

Light availability: A light stress index was generated to represent the relative loss of light available during dredging, and was calculated as $1-(DLI/30)^{1/3}$, where DLI is the daily light integral value and the value of 30 was chosen to represent the maximum sub-surface DLI (actual maximum observed values were ~28 DLI at this location). This relative DLI (DLI/30) was subtracted from one to ensure that stress is represented as increasing values of the Light stress index, such that high scores represent large amounts of lost light. A cube-root transformation was used to remove severe right skew in relative DLI, ensuring greater spread among higher stress index values and a relatively normal distribution.

Suspended sediments: Data on in-situ suspended sediment concentrations were not available consistently throughout the dredging activity near Barrow Island. However, data on turbidity (measured as NTU readings) were available, and while these do not correspond to explicit units of suspended sediment concentrations, they are strongly correlated (Macdonald et al. 2013), and should be sufficient to represent suspended sediment related stressors. Data used consisted of daily mean NTU which were $\log_{10}(NTU+1)$ transformed for further analysis because these are typically highly skewed in dredging programs (Jones et al. 2015a), covering several orders of magnitude, making analysis on a log scale appropriate.

Sediment covering: Deposition was used as an index of stress associated with sediment covering, and was based on surface sediment density as calculated from sideways and upwards facing optical fibre backscatter (OBS) nephelometers and then converted to settled sediment surface density (SSSD, in units of mg cm^{-2}) by laboratory calibration (Ridd et al. 2001) and aggregated to a daily integral (mg $cm^{-2}d^{-1}$). Daily SSSD values were square-root transformed to remove severe right skew.

Relative levels of exposure to each of these three proximal stress indices were considered for each of three exposure dimensions (intensity, frequency and duration) (Table S1.1).

Intensity: Intensity of exposure was captured as the worst case value of each proximal stress metric experienced at a site. Across each daily time series at a site, worst case values were determined from the raw daily mean for each proximal stress index. Daily means can be highly variable from one time to the next. Such short term intensity exposure may be relevant to coral mortality, and we have thus included daily means in our candidate set of water quality metrics. However, daily running means provide a way of capturing these intensity values smoothed across broader (and potentially more relevant) time scales (Table S1.1)(Jones et al. 2015a). Running means were calculated as the average of the previous N_T data points, where N_T is the number of samples in the T day mean. For example, for a 7 day running mean (T=7), N_T = 7 as there are 7 daily values included. The T day running mean at a point in time t is given by equation 1:

$$\bar{x}_T(t) = \frac{1}{N_T}\sum_{i=1}^{N_T} x_i\,(t) \qquad \text{(eqn 1)}$$

where $\bar{x}_{T)}(t)$ is the mean calculated over the previous T days of the data from time t-T to time t days, 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. In R, 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 & Grothendieck 2005) then applying the runmean function from the caTools (Tuszynski 2013) library. Rather than defining arbitrarily the time scale of relevance to corals, we explored worst case values of daily

running means calculated using increasing time spans (1 to 60 days), yielding intensity metrics capturing a range of temporal scales of exposure.

Frequency: The frequency exposure dimensions were captured as the percentage of days (within the 203 day pre-bleaching dredge phase) exceeding baseline conditions for each proximal stress metric (Table S1.1). We examined a range of baseline percentile values (95th, 90th and 80th) aimed at capturing decreasing extremes of baseline conditions, in order to avoid arbitrary decisions on the appropriate percentile value deemed relevant to coral mortality (Table S1.2). Because the baseline time series for some sites was quite short, baseline conditions for defining exceedance were calculated as global values, with a single value used across all sites (see Table S1.2).

Duration: The duration exposure dimension was captured as worst case number of consecutive days exceeding baseline conditions (as for Frequency, above).

Across all three proximal stress indices and the three dimensions as total of 33 possible exposure metrics were calculated and used for relating dredging related pressures to coral mortality and for consideration as possible threshold values.

Table S1.1. Water quality exposure metrics derived for relating to coral mortality. Exposure metrics were derived using three proximal stress indices (Light stress, Turbidity and Deposition) that represent the three key cause-effect pathways for impacts of dredging on corals. Exposure metrics were also derived to capture all three exposure dimensions, including intensity (running mean), frequency (percentage exceedance), and duration (consecutive exceedance).

Type	Name	Calculation/description
A) Light stress	Light stress (-- day running mean)	Maximum observed running mean Light stress [$1-(DLI/30)^{1/3}$] for the pre-bleaching dredging phase. Running means were calculated across time scales of 1, 7, 14, 30 and 60 days.
	Light stress exceedance (--th baseline percentile)	The percentage of pre-bleach dredge phase days where Light Stress values exceed global baseline values. Global baseline values calculated as 95th, 90th and 80th percentiles for all baseline data, across all sites.
	Consecutive light stress (--th baseline percentile)	The maximum number of consecutive pre-bleach dredge phase days where Light Stress values exceed global baseline values. Global baseline values calculated as 95th, 90th and 80th percentiles for all baseline data, across all sites.
B) Turbidity	Turbidity (-- day running mean)	Maximum observed running mean turbidity (log10 NTU + 1) for the pre-bleaching dredging phase. Running means were calculated across time scales of 1, 7, 14, 30 and 60 days.
	Turbidity exceedance (--th baseline percentile)	The percentage of pre-bleach dredge phase days where mean daily log10(NTU+1) values exceed global baseline values. Global baseline values calculated as 95th, 90th and 80th percentiles for all baseline data, across all sites.
	Consecutive turbidity	The maximum number of consecutive pre-bleach dredge phase days where turbidity values exceed global baseline values. Global baseline values calculated as 95th, 90th and 80th percentiles for all baseline data, across all sites.
C) Deposition	Deposition (-- day running mean)	Maximum observed running mean deposition [sqrt(SSSD)] for the pre-bleaching dredging phase. Running means were calculated across time scales of 1, 7, 14, 30 and 60 days.
	Deposition exceedance (--th baseline percentile)	The percentage of pre-bleach dredge phase days where deposition values exceed global baseline values. Global baseline values calculated as 95th, 90th and 80th percentiles for all baseline data, across all sites.
	Consecutive deposition (--th baseline percentile)	The maximum number of consecutive pre-bleach dredge phase days where deposition values exceed global baseline values. Global baseline values calculated as 95th, 90th and 80th percentiles for all baseline data, across all sites.

Table S1.2. Global baseline percentile values used for calculating proportional and consecutive exceedance exposure metrics.

Percentile	Light stress $1-(DLI/30)^{1/3}$	DLI	Turbidity log_{10} (NTU+1)	NTU	Deposition Sqrt(SSSD)	SSSD
80%	0.58	2.2	0.45	1.8	0.19	0.04
90%	0.66	1.2	0.62	3.2	0.26	0.07
95%	0.73	0.6	0.76	4.8	0.43	0.19

Table S1.3. Site level data for 33 exposure metrics (A-C) and probability of non-zero coral mortality relative to controls (D, seep Appendix 2) summarized for the first 203 days of dredging at Barrow Island (19th May 2010 – 8th Dec 2010). Exposure metrics were derived for 3 proximal stress indices (A. Light stress [1-(DLI/30)$^{1/3}$], B. Turbidity (log10 NTU + 1), and C. Deposition [sqrt(SSSD)]) for three exposure dimensions: running mean – worst case (100th percentile) running mean across 1, 7, 14, 30 and 60 day time scales; consecutive exceedance –worst case (100th percentile) number of consecutive days exceeding the 80th, 90th and 95th percentiles; and percentage exceedance – the percentage of days exceeding the 80th, 90th and 95th percentiles. Cell colours indicate the relative intensity for each metric across the sites, with dark red indicate high stress (A-C) or mortality (D) and dark green indicating low stress (A-C) or mortality (D).

	Site	AHC	ANT	BAT	DIW	DUG	ELS	LNG0	LNG1	LNG2	LNG3	LNGA	LNGB	LNGC	MOF1	MOF3	MOFA	MOFB	MOFC	REFN	REFS	SBS	TR
A. Light stress [1-(DLI/30)1/3]																							
running mean	1 d	0.73	0.64	0.66	0.56	0.83	0.70	0.98	0.99	0.87	0.96	0.92	0.99	0.87	0.88	0.83	0.95	0.82	0.83	0.68	0.78	0.68	0.97
	7 d	0.62	0.46	0.58	0.35	0.73	0.54	0.76	0.83	0.75	0.74	0.83	0.82	0.79	0.70	0.73	0.73	0.71	0.71	0.64	0.56	0.52	0.75
	14 d	0.57	0.43	0.53	0.34	0.66	0.52	0.73	0.79	0.72	0.68	0.77	0.77	0.75	0.68	0.71	0.70	0.69	0.68	0.64	0.55	0.50	0.69
	30 d	0.56	0.43	0.52	0.32	0.64	0.52	0.70	0.77	0.70	0.63	0.75	0.72	0.74	0.65	0.65	0.64	0.61	0.62	0.64	0.53	0.45	0.69
	60 d	0.55	0.43	0.49	0.30	0.60	0.52	0.67	0.72	0.66	0.61	0.72	0.70	0.73	0.63	0.60	0.59	0.60	0.58	0.58	0.50	0.45	0.64
consecutive exceedance	>80th (0.58); <2.2 DLI	4	1	2	0	10	2	27	38	37	11	27	20	66	23	15	10	15	14	3	2	1	10
	>90th (0.66); <1.2 DLI	3	0	1	0	3	1	8	23	20	6	13	10	17	4	5	5	7	3	2	2	1	5
	>95th (0.73); <0.6 DLI	1	0	0	0	3	0	5	13	4	2	6	10	8	2	3	3	1	2	0	1	0	4
percentage exceedance	>80th (0.58); <2.2 DLI	13.9	1.0	3.0	0.0	18.8	3.0	59.4	55.0	37.6	25.7	73.3	59.9	66.8	39.1	19.8	28.7	29.2	27.2	5.4	1.5	2.0	14.4
	>90th (0.66); <1.2 DLI	3.5	0.0	0.5	0.0	11.4	1.0	27.7	32.7	21.8	12.9	45.0	35.6	43.1	12.9	9.4	15.8	11.4	8.4	2.5	1.0	0.5	8.9
	>95th (0.73); <0.6 DLI	0.5	0.0	0.0	0.0	6.4	0.0	10.4	16.8	8.4	5.0	18.8	19.8	22.8	3.0	4.0	5.9	3.5	4.0	0.0	0.5	0.0	5.4
B. Turbidity (log10 NTU + 1)																							
running mean	1 d	0.93	0.63	0.75	0.73	1.35	0.72	1.58	1.52	1.40	1.52	1.70	1.51	1.37	1.22	1.38	1.80	1.37	1.51	0.93	0.99	0.93	1.52
	7 d	0.77	0.51	0.57	0.57	0.87	0.50	1.20	1.06	0.95	0.96	1.24	1.07	1.04	0.94	1.00	1.25	0.98	1.14	0.72	0.59	0.69	1.12
	14 d	0.63	0.50	0.48	0.47	0.74	0.40	1.03	0.90	0.87	0.82	1.09	0.90	0.97	0.86	0.87	1.16	0.89	1.03	0.62	0.57	0.64	0.96
	30 d	0.50	0.49	0.48	0.37	0.65	0.38	0.89	0.81	0.84	0.71	0.91	0.77	0.78	0.69	0.74	0.97	0.72	0.80	0.56	0.53	0.58	0.96
	60 d	0.45	0.48	0.41	0.33	0.55	0.35	0.78	0.69	0.79	0.66	0.79	0.65	0.72	0.59	0.66	0.91	0.64	0.74	0.55	0.48	0.58	0.83
consecutive exceedance	>80th (0.45); >1.8 NTU	7	14	5	7	15	4	70	24	53	34	56	20	19	17	20	59	29	23	34	7	18	16
	>90th (0.62); >3.2 NTU	4	1	2	1	9	2	13	13	21	16	11	10	14	6	16	24	10	11	5	2	4	10
	>95th (0.76); >4.8 NTU	2	0	0	0	4	0	10	5	10	6	9	10	13	3	5	16	5	5	3	2	2	5
percentage exceedance	>80th (0.45); >1.8 NTU	18.8	17.3	10.9	4.5	30.2	4.5	68.8	61.9	63.4	51.0	62.4	56.9	47.0	55.4	51.0	68.3	49.0	57.9	24.8	19.3	29.7	37.6
	>90th (0.62); >3.2 NTU	3.0	0.5	1.0	1.0	15.8	2.0	40.6	25.2	33.7	27.2	29.7	27.2	23.3	22.3	22.8	49.0	15.8	27.7	3.5	3.5	9.9	20.3
	>95th (0.76); >4.8 NTU	1.0	0.0	0.0	0.0	9.9	0.0	23.8	14.9	21.3	15.8	16.8	13.4	16.8	9.9	13.9	32.7	8.4	10.9	1.5	1.5	3.5	12.9

C. Deposition [sqrt(SSSD)]																							
running mean	1 d	0.59	0.41	0.98	5.15	0.82	0.55	11.69	2.86	3.15	1.61	4.39	1.61	2.31	4.32	6.63	6.71	3.78	3.96	0.61	0.51	0.44	1.03
	7 d	0.53	0.09	0.21	1.36	0.33	0.36	2.60	2.86	0.75	1.61	1.35	1.61	0.92	1.39	2.60	3.70	1.75	1.98	0.20	0.18	0.15	0.52
	14 d	0.53	0.06	0.12	1.36	0.33	0.36	1.73	2.13	0.50	0.74	1.00	1.10	0.85	1.08	2.60	3.70	0.94	1.32	0.13	0.16	0.15	0.52
	30 d	0.43	0.05	0.09	0.54	0.20	0.09	1.09	1.56	0.40	0.43	0.84	0.64	0.85	1.11	1.13	1.37	0.66	0.86	0.09	0.16	0.15	0.19
	60 d	0.21	0.07	0.08	0.24	0.17	0.06	0.68	0.47	0.32	0.35	0.70	0.33	0.41	0.77	0.48	1.02	0.44	0.84	0.09	0.13	0.15	0.12
consecutive exceedance	>80th (0.19); >0.04 SSSD	4	1	1	1	8	2	11	7	7	5	13	9	15	40	12	13	9	24	2	2	2	3
	>90th (0.26); >0.07 SSSD	2	1	1	1	2	2	10	6	6	5	13	9	15	23	7	12	6	24	2	1	1	2
	>95th (0.43); >0.19 SSSD	2	0	1	1	1	2	6	4	2	2	10	4	9	9	4	7	6	12	1	1	1	1
percentage exceedance	>80th (0.19); >0.04 SSSD	11.9	0.5	2.5	1.5	6.9	1.0	40.6	27.7	25.7	25.2	36.1	15.3	33.7	42.6	35.1	37.1	26.2	28.7	1.5	2.0	1.5	5.9
	>90th (0.26); >0.07 SSSD	6.4	0.5	0.5	0.5	2.5	1.0	29.2	23.8	15.8	16.8	29.2	10.4	25.2	38.1	27.7	31.2	19.8	22.8	1.5	1.0	0.5	3.0
	>95th (0.43); >0.19 SSSD	1.5	0.0	0.5	0.5	1.0	1.0	16.8	16.8	6.4	6.4	22.8	5.4	12.4	24.3	13.9	21.8	13.4	18.3	0.5	0.5	0.5	2.5
D. Probability of non-zero coral mortality																							
	Acropora Br	0.00	0.00	NA	0.00	0.02	0.00	NA	NA	NA	0.01	NA	NA	0.33	0.47	0.79	0.83	0.44	0.03	0.00	0.00	0.00	0.18
	Faviidae	0.00	0.00	0.19	0.00	0.37	0.00	NA	0.65	0.02	NA	0.29	NA	NA	0.85	0.04	0.27	0.14	0.68	0.00	0.00	0.00	0.01
	Pocilloporidae	0.00	NA	0.00	NA	0.00	0.00	0.91	0.23	0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	0.00	NA	0.00	NA
	Poritidae Ms	0.00	NA	0.00	NA	0.00	0.00	0.75	0.53	0.03	0.05	0.88	0.30	0.75	0.08	NA	0.97	0.61	0.12	0.00	0.00	0.00	0.00
	Worst case	0.00	0.00	0.19	0.00	0.37	0.00	0.91	0.65	0.64	0.05	0.88	0.30	0.75	0.85	0.79	0.97	0.61	0.68	0.00	0.00	0.00	0.18

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Appendix S2. Detailed methods - Probability of coral mortality

Collection and processing of live coral head data

Live coral cover data were recorded from individual coral heads that were monitored approximately every fortnight throughout the dredging phase. Data were obtained using Coral Point Count with Excel Extensions (CPCe, (Kohler & Gill 2006)) based on repeated photographs of individual coral heads, taken periodically throughout the study period. Some coral heads were excluded from the analysis if they had been dislodged (identified by a field labelled "Dislodged.Issue") or if there was some other issue with their photograph (identified by a field labelled "Photo.Issue"). In addition, coral heads were only used in the analysis where there were at least 8 observations of the coral head through time (including measurements in the baseline period), and where more than half the coral was scored as "live" at the start of dredging.

Each individual coral head percentage of live cover time series was fitted using statistical modelling methods in order to smooth across random variation in scoring and also to allow interpolation of the data across the time series so that it could be more easily coupled with the water quality datasets at multiple scales. Because of the highly variable nature of the individual coral head time series, considerable work was invested in developing an optimal algorithm capable of accurately capturing a range of possible coral health trajectories, from slightly non-linear curves to extremely rapid declines. Here we have used a Generalised Additive Model (gam) from the R (R Core Team 2016) package mgcv (Wood 2006) that uses a cubic regression spline to fit an optimal smooth across the time series, with events greater than 50% loss fitted as an additional fixed term in the model. This modelling approach allows gradual changes in the coral cover data to be captured as smooths, whilst also accurately capturing sudden declines in cover (rapid loss in coral cover between consecutive sampling periods). Data were modelled using a beta distribution on the proportion of live coral cover. Coral cover scores representing sudden dips or peaks of coral cover representing more than 25% (ie the cover dropped by 25% cover at one time step and then returned by 25% cover in the next) were excluded from the fit to avoid biasing coral cover data and calculated mortality rates where live coral was only temporarily covered by sediments or was in some other way temporarily obscured (Fig S2.1).

After data cleaning procedures were carried out, the number of coral heads monitored at individual sites around Barrow Island ranged from 18 up to 67. Coral heads were divided into unique morphology and family level groups, with the exception of the genus *Acropora*, which was retained at the genera level, and the families Mussidae and Faviidae, which were combined as single group of Favid like corals. Morphology types included "branching" (br, which included branching and corymbose corals), "tabulate" (Tb), "encrusting" (En) and "massive" (Ms) (Table S2.1). Of these four family/morphology groups had sufficient replicate colonies > 3) across enough control sites (>2) for analysis and included: Branching *Acropora* (251 colonies total), Branching Pocilloporidae (58 colonies total), Massive Faviidae (134 colonies total), Massive Poritidae (510 colonies total) (Table S2.1).

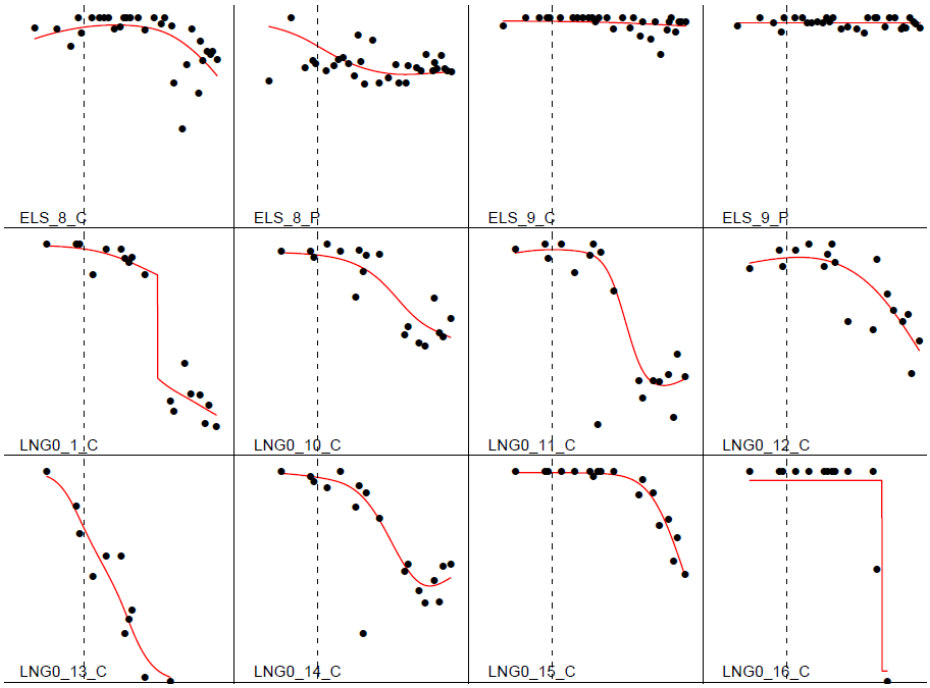

Fig. S2.1. Example coral head trajectories for site LNG0 for the Barrow Island project (black circles), showing modelled GAM fits (red line).

Table S2.1. Total number of coral heads for which there were usable data for analysis of coral mortality at Barrow Island. Corals were divided into 15 morphology/Family groups. Corals were grouped at the family level with the exception of the genus *Acropora* (which was retained at the genus level) and the Faviidae/Mussidae (which were included as a single group labelled "Faviidae"). Morphology types were: "branching" (br, which included both branching and corymbose forms), "tabulate" (Tb), "encrusting" (En) and "massive" (Ms). Where no morphology was recorded next to a family/group this means that all colonies in that family were of a single morphology type. Site codes are shown and correspond to those identified on Fig 1 of the main text. Dredging status indicates which sites were controls, and which sites were considered as potential impact sites for the purpose of the analysis. All control sites were used as controls for each potential impact site.

Site	Dredging status	*Acropora* Br	*Acropora* Tb	Acroporidae	Agariciidae	Dendrophylliidae	Faviidae	Fungiidae	Merulinidae	Oculinidae	Pectiniidae	Pocilloporidae	Poritidae Br	Poritidae En	Poritidae Ms	Siderastreidae	Total
AHC	C	7	0	0	0	1	3	0	0	2	0	3	19	0	22	0	57
ANT	C	3	0	0	0	0	6	0	3	2	0	2	0	0	2	0	18
BAT	C	1	0	0	0	0	14	0	3	0	0	5	13	1	27	0	64
DIW	C	30	0	0	0	0	9	0	1	1	0	0	0	0	0	0	41
DUG	PI	5	0	5	5	0	10	0	1	1	4	3	4	1	25	0	64
ELS	C	26	0	0	0	0	4	0	1	0	0	6	0	0	29	0	66
LNG0	PI	1	0	0	0	0	0	0	0	0	0	5	5	0	32	0	43
LNG1	PI	2	0	0	0	0	3	0	1	0	0	5	1	0	31	0	43
LNG2	PI	1	0	0	0	2	8	0	0	0	0	5	0	0	29	0	45
LNG3	PI	10	0	0	0	2	1	0	1	0	0	2	5	0	24	1	46
LNGA	PI	0	0	0	0	4	3	0	0	0	0	1	0	0	34	0	42
LNGB	PI	0	0	0	0	0	0	0	0	0	0	2	2	0	42	0	46
LNGC	PI	7	0	0	0	2	0	0	0	0	0	2	12	0	22	0	45
MOF1	PI	30	0	2	0	0	9	0	0	0	1	0	1	0	19	0	62
MOF3	PI	33	0	1	0	0	6	1	1	0	1	1	0	0	1	0	45
MOFA	PI	6	0	2	0	2	16	1	2	0	4	0	0	0	24	0	57
MOFB	PI	26	1	0	0	1	6	0	1	0	2	0	0	0	22	0	59
MOFC	PI	22	1	1	1	0	5	0	0	0	0	0	0	0	35	0	65
REFN	C	4	0	0	0	0	4	0	0	10	0	6	3	0	31	0	58
REFS	C	8	0	0	6	0	15	0	1	0	0	2	6	0	29	0	67
SBS	C	17	1	9	1	0	3	1	0	0	0	7	1	1	22	0	63
TR	PI	12	0	0	6	0	9	1	2	0	4	1	3	0	8	0	46
Total		251	3	20	19	14	134	4	18	16	16	58	75	3	510	1	1142

Calculating probability of mortality for potential impact sites

In order to examine how water quality conditions during dredging relate to coral mortality it was necessary to factor out natural mortality that occurs even in the absence of dredging related pressures, and calculate the likelihood that there was in fact elevated mortality at any of the given potential impact sites. Here we use Bayesian hierarchical models to calculate the posterior probability of non-zero coral mortality at each potential impact sites. All potential impact sites (those inside the maximum bounds of observed impacts on water quality, open circles, Fig. 1) were individual statistically examined for evidence of increased mortality relative to the designated control sites (those outside the maximum observed bounds of observed impacts on water quality, closed circles, Fig. 1).

We used a Bayesian statistical approach to calculate the probability that a potential impact site experienced mortality greater than the set of control sites. This was done because the Bayesian MCMC statistical fitting approach provides an estimate of the statistical distribution around model fits that can be used to assess such probabilities explicitly. As the response variable for the Bayesian analysis to examine coral mortality, we used the difference in the percentage coral cover measurements (ΔC) for each colony from the start of dredging (startC) to the end of the dredge period (EndC), where ΔC takes values of between 0 (no partial mortality) and 100 (complete coral head mortality, for a coral head scoring 100 percentage cover) and represents total percentage loss of live coral occurring for each colony over the dredge period.

$$\Delta C = StartC - EndC \qquad \text{eqn 1}$$

We assumed that the percentage live coral loss (ΔC) was a binomial random variable, dependent on percentage cover at the start of dredging (T_i) and probability P_i (see eqn 2). This was modelled via a logit link as a function of the fixed factor (CvI; distinguishing between the control locations and the specific potential impact location being examined) and a random site level offset a_{ji} to account for the non-independence of coral heads collected at individual sites (see eqn 3), assumed to be normally distributed with mean of zero and variance σ^2 estimated as a

parameter in the model. All sites outside of the potential zone of impact that had more than three coral heads of the relevant taxonomic type were designated as controls. The random effect of site was only based on random variance associated with these control sites (as there was only 1 impact site for each individual site test). Parameters $\beta 0$ and $\beta 1$ are the estimated parameters of the logistic regression (see eqn 2).

$$\Delta C_i \sim \text{binomial}(T_i, P_i) \qquad \text{eqn 2}$$

$$\text{logit}(P_i) \sim \beta_0 + \beta_1 \text{ x CvI} + a_{ji} \qquad \text{eqn 3}$$

Random normal diffuse non-informative priors were used for the parameter estimates for β_0 and β_1, with half-Cauchy priors (Gelman 2006) used for the random site variance σ^2. Models were fit in Jags (Plummer 2003), using the R2jags package (Su & Yajima 2015) in R (R Core Team 2014). To aid convergence starting values of β_0 and β_1 were based on the equivalent model fit using glmer from the lme4 library (Bates & Maechler 2010) with starting values for the random site level deviations taken from a normal distribution with mean of zero and standard deviation of 0.001. We ran five MCMC chains each with 100,000 iterations and 30,000 iterations were discarded as 'burn-in'. Chain mixing was assessed visually using trace plots, and found to be acceptable.

A posterior probability distribution of the relative proportional mortality (ΔM) for each sites was then calculated from the MCMC iterations as the inverse logit of the difference in the model estimated proportional coral loss at the impact location being examined (I_m) minus the modelled mortality at the control locations (C_m) (See Fig S2.2).

$$\Delta M = I_m - C_m \qquad \text{eqn 4}$$

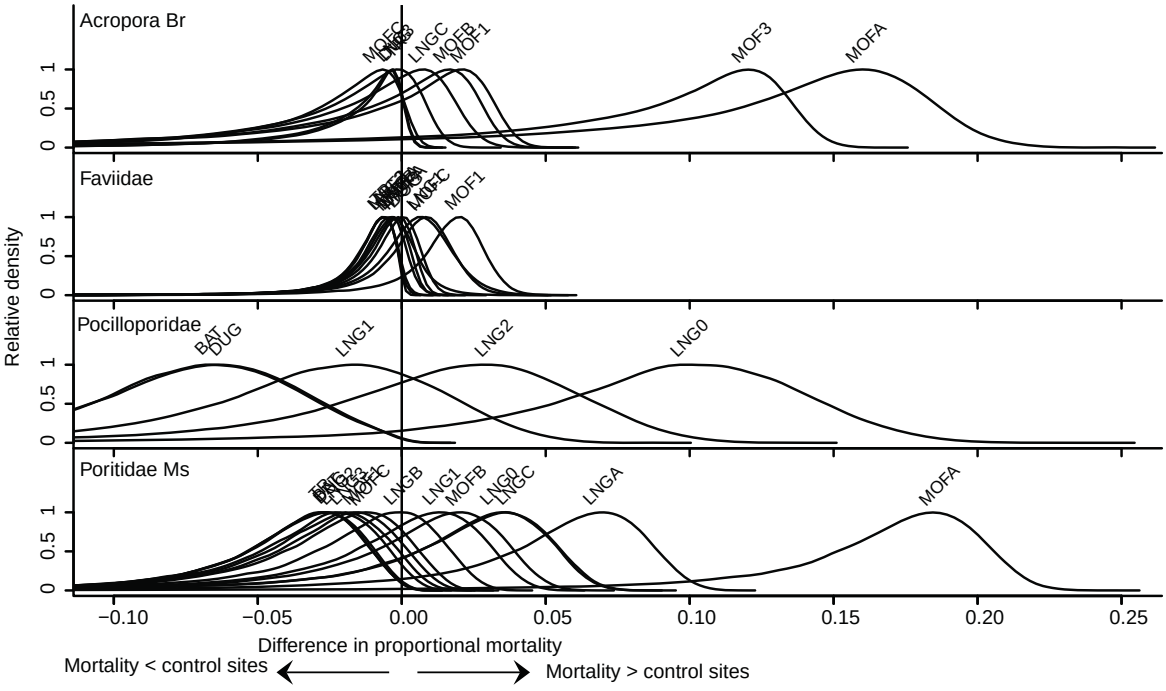

Fig. S2.2. Estimates of dredging related relative coral loss at Barrow Island at potential impact sites for the first 203 d of dredging. Plotted curves represent posterior probability densities of the difference in estimated percentage coral loss at potential impact sites relative to controls, obtained from Bayesian mixed model analysis.

Positive values of ΔM (>0) indicate that mortality at the site of interest was in excess of that observed at control locations, and negative values (<0) indicate that mortality at a site was less than that observed across the control locations. Values near zero indicate that mortality at the impact and control locations are similar. The posterior probability of non-zero mortality was then calculated as the proportion of the MCMC sample of relative proportional mortality (ΔM) that was above zero. Posterior probabilities of non-zero mortality were calculated individually for each potential impact site, for each family/morphology group for which there were sufficient replication (greater than three colonies at the impact site and more than two control locations).

Analyses were run individually for all family/morphology groups for which there were sufficient colonies (>3) at most sites, and at least two control sites. To simplify following analysis based on the probability of non-zero mortality, we used the maximum observed mortality values for any one taxonomic group, under the assumptions that this would provide the most conservative estimates of calculated threshold values best able to protect the coral community as a whole. While averaging across taxa was considered, rather than taking a maximum value, it was decided that this would be inappropriate because for an average to be useful, it should be based on randomly selected taxa representative of the "community" and there is insufficient replication across enough taxa for this to be possible. The different taxa have different levels of statistical power to detect change, but also respond to different aspects of the dredging related pressure. Selecting the worst case mortality across the taxa is conservative, as this would yield thresholds that have the best chance of protecting the community as a whole because mortality for any of the taxa at a site suggests that the conditions at that site are such that it can lead to mortality for at least some part of the coral community. Future work will explore differences in the potential sensitivity across taxa and how these might alter calculated threshold values.

```
X <- model.matrix(~CvI, data = dat.i)
K <- ncol(X)

#Random effects:
re.site  <- as.numeric(factor(dat.i$Site.Code))
Num.site <- length(unique(dat.i$Site.Code))

#Put all required data in a list
win.data <- list(Y        = successes,
             Trials       = trials,
             N            = nrow(dat.i),
             X            = X,
             RDum         = 1-X[,2], # Vector of 1s for the control sites
                                     # so the random effect of site is
                                     # only based on controls
             K            = ncol(X),
             re.site      = re.site,
             Num.site     = Num.site)
#JAGS code
sink("binomialJAGS.txt")
cat("
model{
 #Likelihood
 for (i in 1:N) {
  Y[i]          ~ dbinom(Pi[i],Trials[i])
  logit(Pi[i]) <- eta[i]
  eta[i]     <- inprod(beta[], X[i,]) + b[re.site[i]] * RDum[i]
  }
 #--Priors betas
 for (i in 1:K) { beta[i] ~ dnorm(0, 0.1)}
 #--Priors random effects
 for (i in 1:Num.site) { b[i] ~ dnorm(0, tau_site)}
 #  Priors for sigmas
 tau_site <- 1 / sigma_site^2
 num_site           ~ dnorm(0, 0.0016)              #<----half-Cauchy(25)
 denom_site         ~ dnorm(0, 1)                   #<----half-Cauchy(25)
 sigma_site    <- abs(num_site / denom_site)        #<----half-Cauchy(25)
}
",fill = TRUE)
sink()
#--------------------------------
inits  <- function () {
 list(
   beta  = rnorm(ncol(X), coef(summary(mod.2))[,1], 0),
   b     = rnorm(Num.site, 0, 0.001),
   num_site = rnorm(1, 0, 25),
   denom_site = rnorm(1, 0, 1))  }
params <- c("beta", "b","sigma_site")
load.module("dic")
load.module("glm")
J1 <- jags(data       = win.data,
       inits      = inits,
       parameters = params,
       model      = "binomialJAGS.txt",
       n.thin     = 5,
       n.chains   = 5,
       n.burnin   = 30000,
       n.iter     = 100000)
out <- J1$BUGSoutput
Beta.mcmc <- out$sims.list$beta
eta.mcmc <- model.matrix(~CvI,data=data.frame(CvI=c(0,1)))%*%t(Beta.mcmc)
Pi.mcmc <- exp(eta.mcmc) / (1 + exp(eta.mcmc))
C.mcmc=Pi.mcmc[1,]
I.mcmc=Pi.mcmc[2,]
dif.mcmc=(I.mcmc-C.mcmc)
```

Fig. S2.3. R and R2jags code detailing the analysis of coral head data to estimate the fixed effects of $_{a1}$C and $_{a2}$I, and the corresponding relative change in proportional mortality ΔM = I*m* - C_m (shown as dif.mcmc in the code). In the binomial fit "successes" is the change in percentage of live cover (ΔC) and "trials" is the percentage of live cover of each coral head at the start of dredging. As we only used the change in live cover as our response variable (1 value per coral head), coral heads are our replicates and there are no repeated measures effects, although the random effect of site is captured.

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Appendix S3. Thermal bleaching

At Barrow Island a widespread thermal bleaching event occurred in the summer of 2010-2011 during the dredging period, due to a warm water anomaly (Moore et al. 2012, Ridgway et al. 2016). Data on temperature as well as the extent of coral bleaching were recorded throughout the time series. This information was used to determine what portion of the dredging period was clearly not impacted by the bleaching event.

Methods

Bleaching score data were first smoothed across the individual coral head time series. Statistical fits of percentage bleaching based on GAM smoothing were found to be unsatisfactory because these methods were unable to adequately capture the rapid changes observed in the bleaching time series. Instead smooths were achieved using the function rqss from the quantreg (Koenker 2015) package in R, using the upper 80th percentile values. Interpolated daily bleaching values were calculated as percentages of the live coral tissue, and where the live coral tissue was less than 10% they were discarded to avoid using unreliable proportions (few scoring points would be available over the live coral portion of the coral head).

Temporal patterns in mean bleaching (across all colonies) were examined throughout the time series and the day at which bleaching was first observed was estimated using hierarchical broken stick (aka piecewise) linear regressions implemented using the lme4 package in R (Bates et al. 2015) with site included as a random effect.

Results and Discussion

Time series of temperature and the proportion of bleached live tissue observed on coral heads indicated the occurrence of a substantial bleaching event at Barrow Island (Fig S3.1). Maximum severity of the bleaching event occurred at 269 days into dredging for Barrow Island (solid red vertical line, Fig S3.1), with break-point regression on the pre-peak bleaching data indicating the event started at around 203 days respectively (dashed red vertical line, Fig S3.1). To avoid issues with confounding bleaching effects with dredging effects, thresholds were developed based on this 0-203 day pre bleaching period.

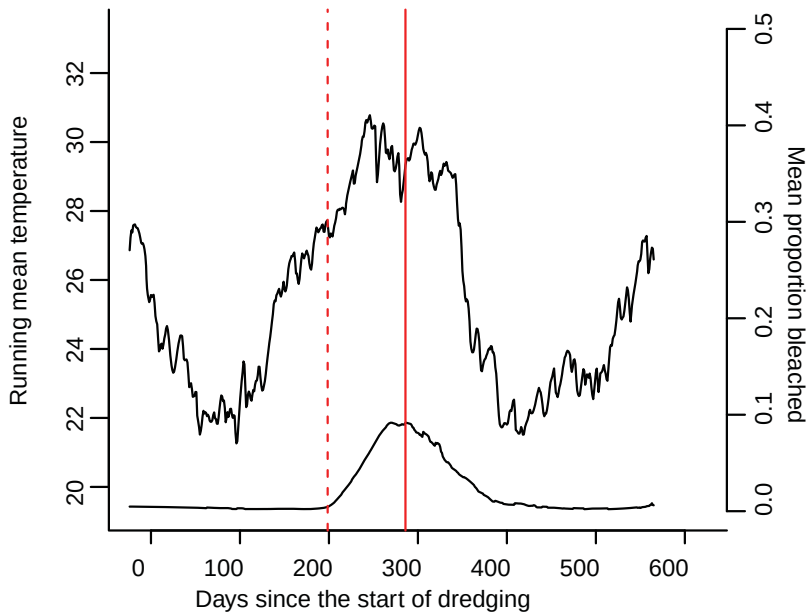

Fig S3.1. The relationship between running mean temperature and coral bleaching at Barrow Island. Shown is the time series of daily mean temperatures (black line, left hand Y-axis), with the mean observed proportion of bleached live tissues on each day (grey line, right hand Y-axis). The dashed vertical red line indicates the time of maximum bleaching, while the dashed red line represents the start of the bleaching event.

References

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Appendix S4. Detailed methods - Threshold development and Receiver operating characteristic (ROC) curves.

Thresholds were developed based on the best candidate metric (based on minimum AICc, (Burnham & Anderson 2002)) for each of the three proximal stress indices (Light stress, Turbidity and Deposition) and for each of the exposure dimensions (running mean, percentage exceedance and consecutive exceedance) (nine in total). We used a formal quantitative decision science approach for threshold development, where receiver operating characteristic (ROC) curves were calculated, and thresholds derived based on hypothetical desired true positive and false positive probabilities (Swets et al. 2000).

Most existing software for calculating ROC curves is based on a binary response variable (ie impacted or not-impacted) rather than a continuous response, such as a posterior probability of non-zero mortality as we have calculated here (see Appendix S1 for details). To accommodate such probabilities we developed a simulation procedure, where ROC curves were calculated by repeatedly allocating sites to either impact, or non-impact, based on their observed posterior probability of mortality. Thus the ROC curves calculated, along with associated statistics and derived thresholds, were represented by 95% probability distributions reflecting this uncertainty in mortality.

Threshold values for each metric were derived as follows: For each interaction of simulation procedure, individual sites were allocated randomly to an impact or non-impact site based on the probability of non-zero mortality at that site. For all control sites, the posterior probability of non-zero mortality were defined as zero (all were by definition considered controls). Based on our treatment of the posterior MCMC sample (where the estimate of each potential impact size is calculated as a difference from the estimated mortality observed across the controls (after factoring out site-level variance; $\Delta M = I_m - C_m$, see Appendix 1) a site with equal mortality to the control sites will have a symmetric distribution around zero. The proportion of the MCMC sample that is above zero for a theoretically "zero" mortality site (i.e. equal to the controls) would be exactly 0.5. As such, it can be argued that there is no evidence that sites of less than a 0.5 probability of non-zero mortality were impacted in any way by the dredging (in fact, if below 0.5 there is evidence that the mortality was less than at the controls). For the purposes of the ROC curve analysis all sites with a probability of non-zero mortality of <0.5 were therefore assigned a probability of zero so that they were always designated as "controls" in the ROC simulation analysis. Where the posterior probability of non-zero mortality is greater than 0.5, this suggest some evidence of mortality at that site, and such sites were included in the simulation study as an "impact" site at a rate proportional to their posterior probability of non-zero mortality (between 0.5 and 1, depending on the site, see Fig 4 in the main text).

Probability density distributions for the metric being examined were constructed individually for the impact and non-impact locations, and saved at each iteration so these could be presented as 95% confidence bands at the end of the simulation. In addition, for each iteration of the simulation, a set of threshold values were generated for the exposure metric being examined. Threshold value selection was included as a random variable in the simulations to avoid creating artificial threshold bounds associated with the arbitrary choice of threshold bins. Threshold values were generated by using the runif function in R to randomly select a single threshold value within each of 15 bins across the range of the observed values. The maximum and minimum of the observed values were also included, to ensure that the full range of possible values was represented at each iteration.

For each simulated threshold value, the proportion of true positives (TP) was calculated as the proportion of "impact" sites that exceeded the threshold value for that iteration. The proportion of false positives (FP) was calculated as the proportion of control sites that exceeded the threshold value for that iteration. As the number of replicate sites was relatively low, fitted non-linear regressions were used to smooth the relationships between both TP and FP values across the range of threshold values. This interpolation was done for each fixed combination of the thresholds used for the other included pressure variable(s). Interpolation was achieved through a custom self-starting non-linear least squares regression algorithm (developed in R, based on the function nls) that selects the best (defined as that model fit with the smallest AICc value) of six possible smoothing relationships. A ROC curves was then constructed for that iteration, based on the smoothed true positive and false positive probabilities.

From each iteration of the ROC curve we derived two potential threshold values, including: a) a "strict" threshold, which represents the threshold value where the true positive probability is 0.95 (a 95% chance of detecting coral loss, when it occurs); and b) a "permissive" threshold, which represents the threshold value where the false positive probability is 0.05 (a 5% chance of generating a false alarm). We also calculated the area under the ROC curve, as a measure of the accuracy of the curve (including 95% confidence limits).

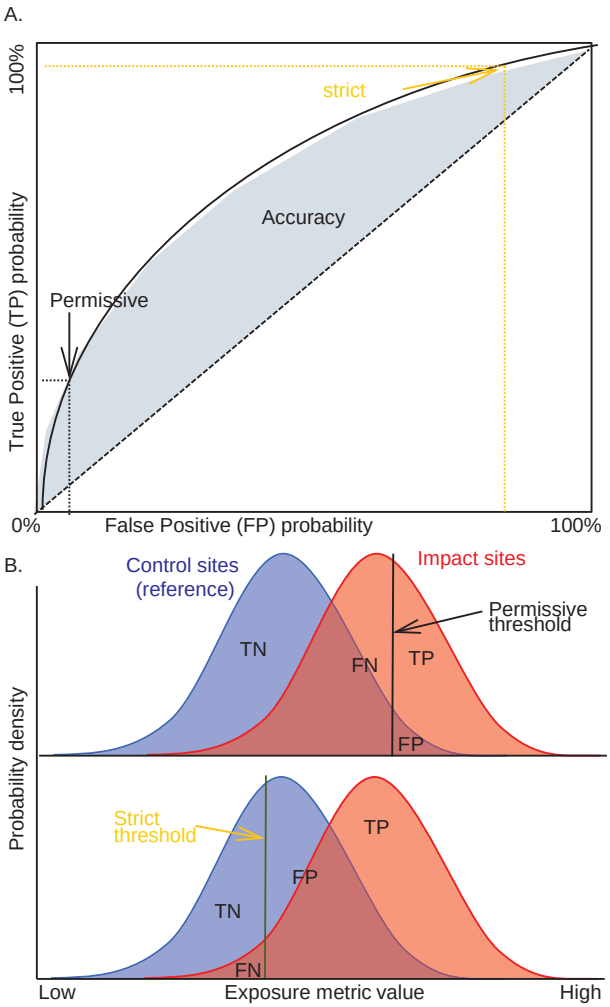

Fig. S4.1. Schematic diagram of a receiver operating characteristics (ROC) curve (A) and corresponding probability density functions for control (blue) and impact (red) sites for a hypothetical exposure metric (B). The ROC curve is constructed by selecting a sequence of thresholds values from the lowest to the highest value of the exposure metric, and plotting the percentage of true positive values (the percentage of impact sites that exceed the threshold, TP) against the percentage of false positive values (the percentage of control sites that exceed the threshold). Two different thresholds can be derived from these curves: Permissive thresholds – where the focus is on minimising the rate of false positives (upper graph, panel B); and Strict thresholds – where the focus is on maximising the rate of true positive (lower graph, panel B). The area under the curve represents a metric of accuracy of the ROC curve (grey shaded area, panel A).

References

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Appendix S5. Detailed results - Dose-response relationships

We examined the dose response relationships between coral mortality and 33 possible exposure metrics based on water quality. These included 5 different running mean values (1, 7, 14, 30 and 60 day) as measures of intensity, the proportion of pre-bleach dredging days exceeding three global baseline percentile values (95th, 90th and 80th) as measures of frequency, and the worst case number of consecutive days exceeding three global baseline percentile values (95th, 90th and 80th) as measures of duration. For each of these three exposure dimensions (IFD) exposure metrics were calculated for each of the three proximal stress indices representing the three key cause-effect pathways for dredging related impacts on coral (Light stress, Turbidity and Deposition). Dose-response relationships were fitted at the site level using a binomial generalized linear model fit using the glm function from package stats in R. All metrics were compared using AICc (Burnham and Anderson 2002) and R^2 values, and estimated parameters reported (Table 6.1). All 33 dose-response relationships are presented in Figs. S5.1-S5.3.

Table S5.1. Model fit statistics for all dose-response relationships between the maximum observed (across taxa) probability of non-zero mortality and 33 derived exposure metrics. Metrics were derived based on three proximal stress indices (Deposition, Turbidity and Light stress) for each of the three exposure dimensions (running mean; percentage exceedance; and consecutive exceedance), and are sorted from lowest to highest AICc, with the best fit relationship for each type of metric highlighted in bold. Predicted probabilities of mortality can be calculated from the parameters *a* and *b*, based on the logistic regression: y = 1/(1+exp(-(-*a*+*b**x))).

Pressure variable	R^2	AICc	Delta AICc	wi	*a*	*b*
Deposition exceedance (base. 80th perc.)	**0.78**	**397**	**0**	**1**	**-3.16**	**0.13**
Deposition exceedance (base. 90th perc.)	0.77	413	16	0	-2.78	0.15
Deposition exceedance (base. 95th perc.)	0.75	447	49.9	0	-2.39	0.22
Deposition (60 day running mean)	**0.69**	**539**	**142**	**0**	**-2.97**	**6.82**
Turbidity exceedance (base. 80th perc.)	**0.68**	**545**	**148**	**0**	**-4.61**	**0.09**
Turbidity (14 day running mean)	**0.68**	**556**	**158**	**0**	**-8.1**	**9.23**
Turbidity exceedance (base. 90th perc.)	0.62	640	243	0	-3.03	0.13
Turbidity (7 day running mean)	0.62	645	247	0	-7.58	7.59
Deposition (30 day running mean)	0.62	647	250	0	-2.69	3.7
Turbidity exceedance (base. 95th perc.)	0.58	695	298	0	-2.59	0.2
Consecutive Deposition (base. 95th perc.)	**0.56**	**732**	**335**	**0**	**-2.17**	**0.45**
Light stress (14 day running mean)	**0.55**	**741**	**344**	**0**	**-11.9**	**17.2**
Turbidity (30 day running mean)	0.54	766	369	0	-6.84	8.99
Turbidity (1 day running mean)	0.53	774	377	0	-6.58	4.71
Consecutive Deposition (base. 80th perc.)	0.53	783	386	0	-2.26	0.23
Deposition (1 day running mean)	0.52	795	398	0	-2.15	0.61
Consecutive Deposition (base. 90th perc.)	0.51	810	413	0	-1.97	0.24
Light stress (7 day running mean)	0.49	836	439	0	-10.4	14.3
Turbidity (60 day running mean)	0.49	842	444	0	-6.59	9.65
Light stress exceedance (base. 80th perc.)	**0.48**	**850**	**453**	**0**	**-2.05**	**0.06**
Deposition (7 day running mean)	0.46	883	486	0	-2.12	1.35
Consecutive Turbidity (base. 95th perc.)	**0.44**	**905**	**508**	**0**	**-2.08**	**0.31**
Light stress (30 day running mean)	0.44	907	509	0	-9.54	14.4
Consecutive Turbidity (base. 90th perc.)	0.44	907	509	0	-2.49	0.21
Deposition (14 day running mean)	0.43	922	525	0	-1.98	1.66
Light stress (60 day running mean)	0.42	933	536	0	-9.23	14.6
Consecutive Light stress (base. 80th perc.)	**0.4**	**970**	**572**	**0**	**-1.73**	**0.09**
Light stress exceedance (base. 90th perc.)	0.37	1022	624	0	-1.57	0.08
Light stress (1 day running mean)	0.36	1034	637	0	-8.26	9.35
Consecutive Turbidity (base. 80th perc.)	0.35	1047	650	0	-1.96	0.06
Light stress exceedance (base. 95th perc.)	0.25	1189	792	0	-1.22	0.13
Consecutive Light stress (base. 90th perc.)	0.25	1202	805	0	-1.32	0.15
Consecutive Light stress (base. 95th perc.)	0.2	1268	870	0	-1.16	0.24
Null model	0	1567	1170	0	-0.4	

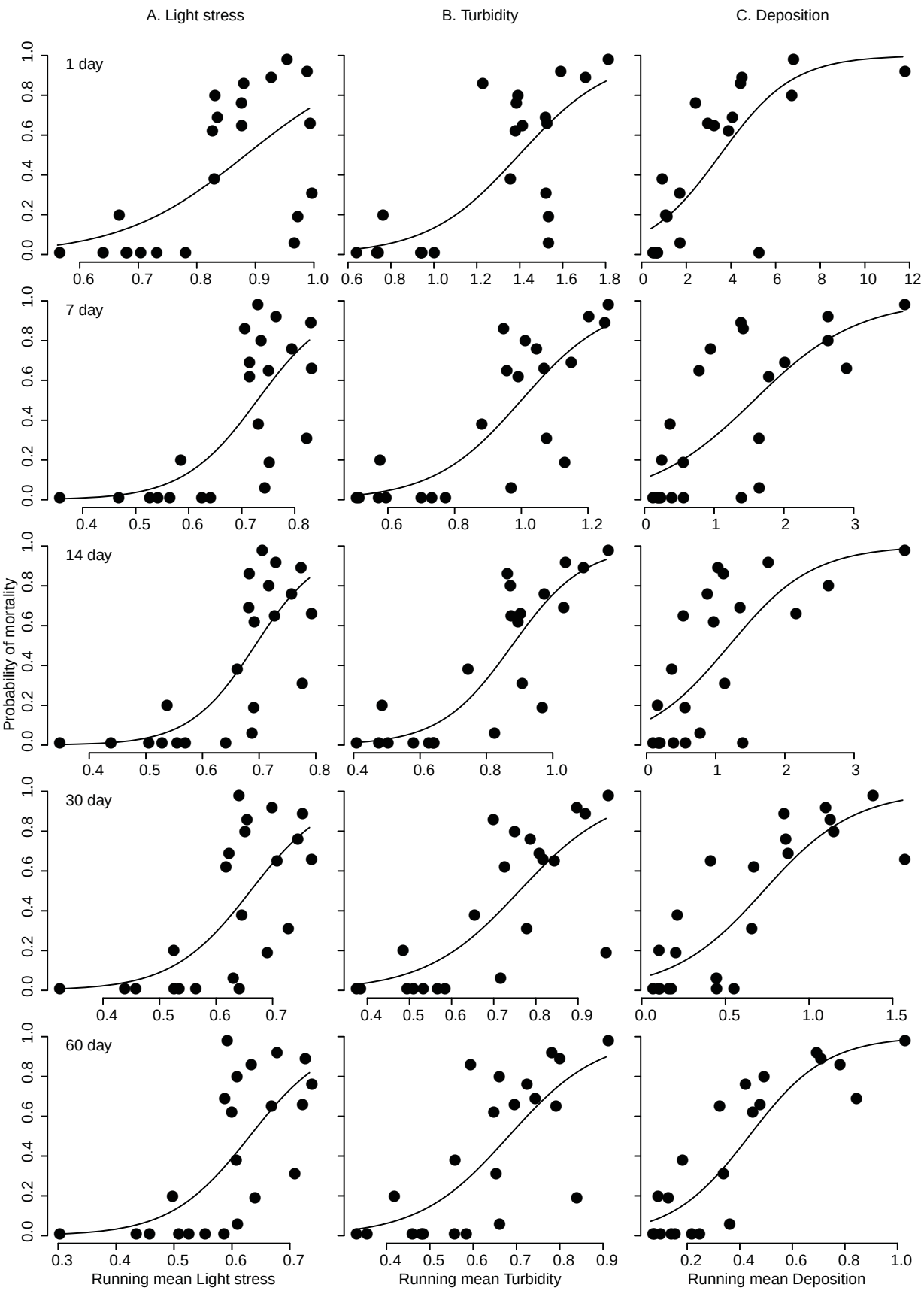

Fig S5.1. Dose response relationships between the probability of coral mortality across 22 sites at Barrow Island, and worst case (over the 203 pre-bleach during dredging days) running mean values for Light stress, Turbidity and Deposition. Running means are based on daily means, and cover time spans of 1 day (a single daily mean) to 60 days (the mean of the 60 preceding daily means). Running means for Turbidity are based on log10(NTU+1) daily means, and Deposition represents sqrt(SSSD).

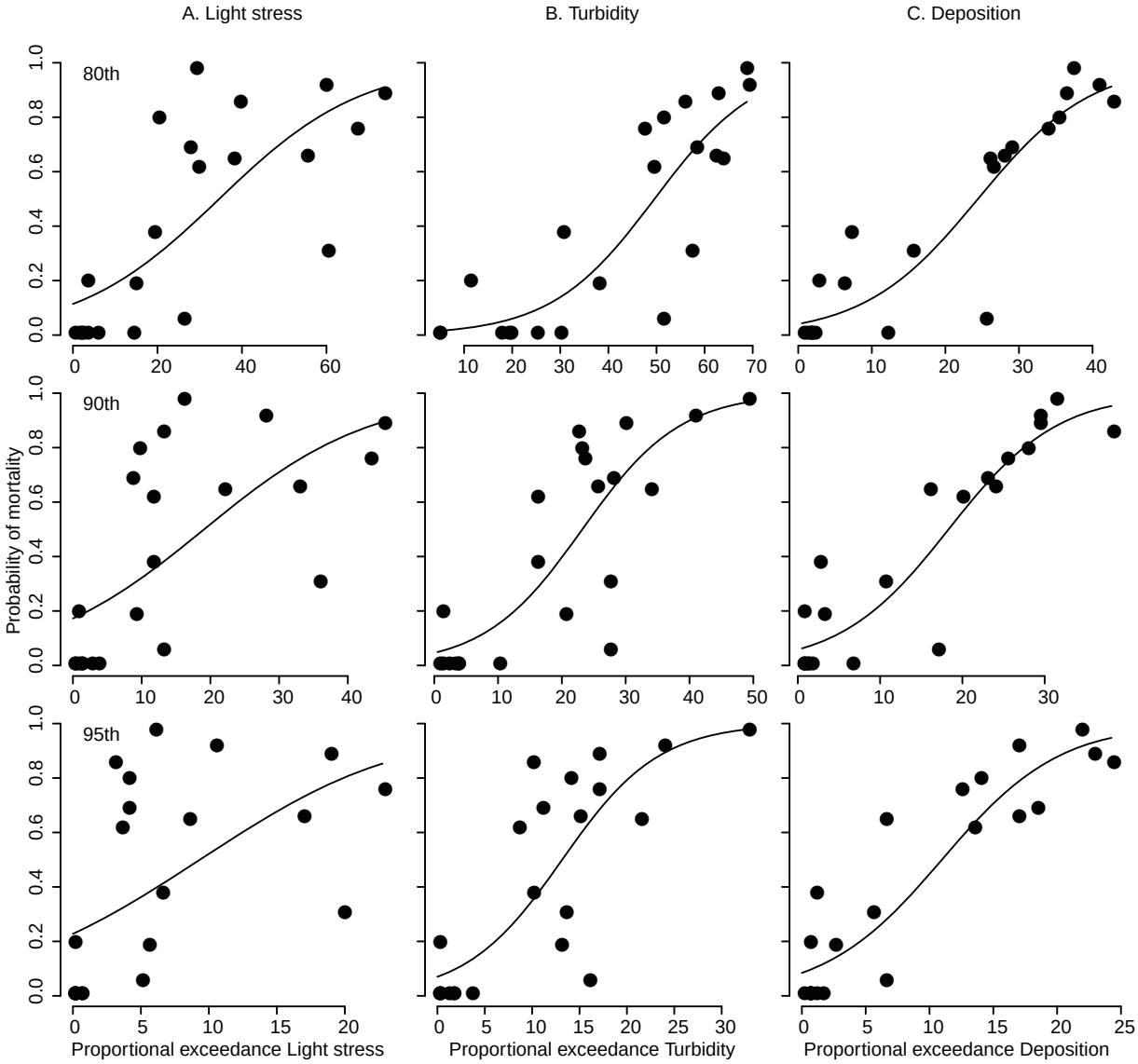

Fig S5.2. Dose response relationships between the probability of coral mortality across 22 sites at Barrow Island, and proportional exceedance values for Light stress, Turbidity and Deposition. Proportional exceedances were calculated for 3 different global baseline percentiles (80th, 90th, 95th) and represent the proportion of days each percentile based cut-off value were exceeded out of the 203 pre-bleach during dredging days.

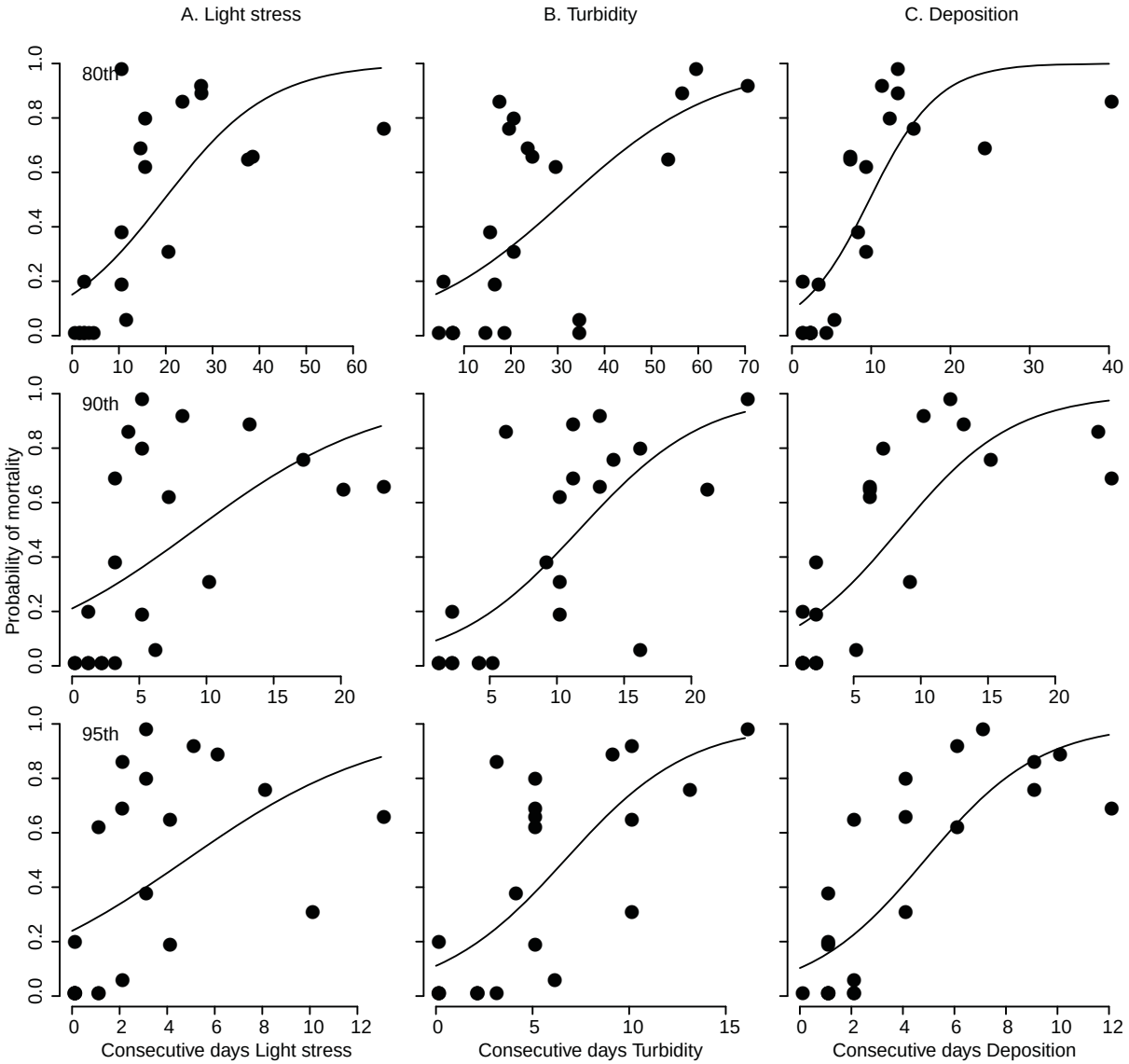

Fig S5.3. Dose response relationships between the probability of coral mortality across 22 sites at Barrow Island, and the worst case (over the 203 pre-bleach during dredging days) case consecutive number of days exceeding the 3 different global baseline percentiles (80th, 90th, 95th) for Light stress, Turbidity and Deposition

Appendix S6. Detailed results - Water quality thresholds and ROC curves

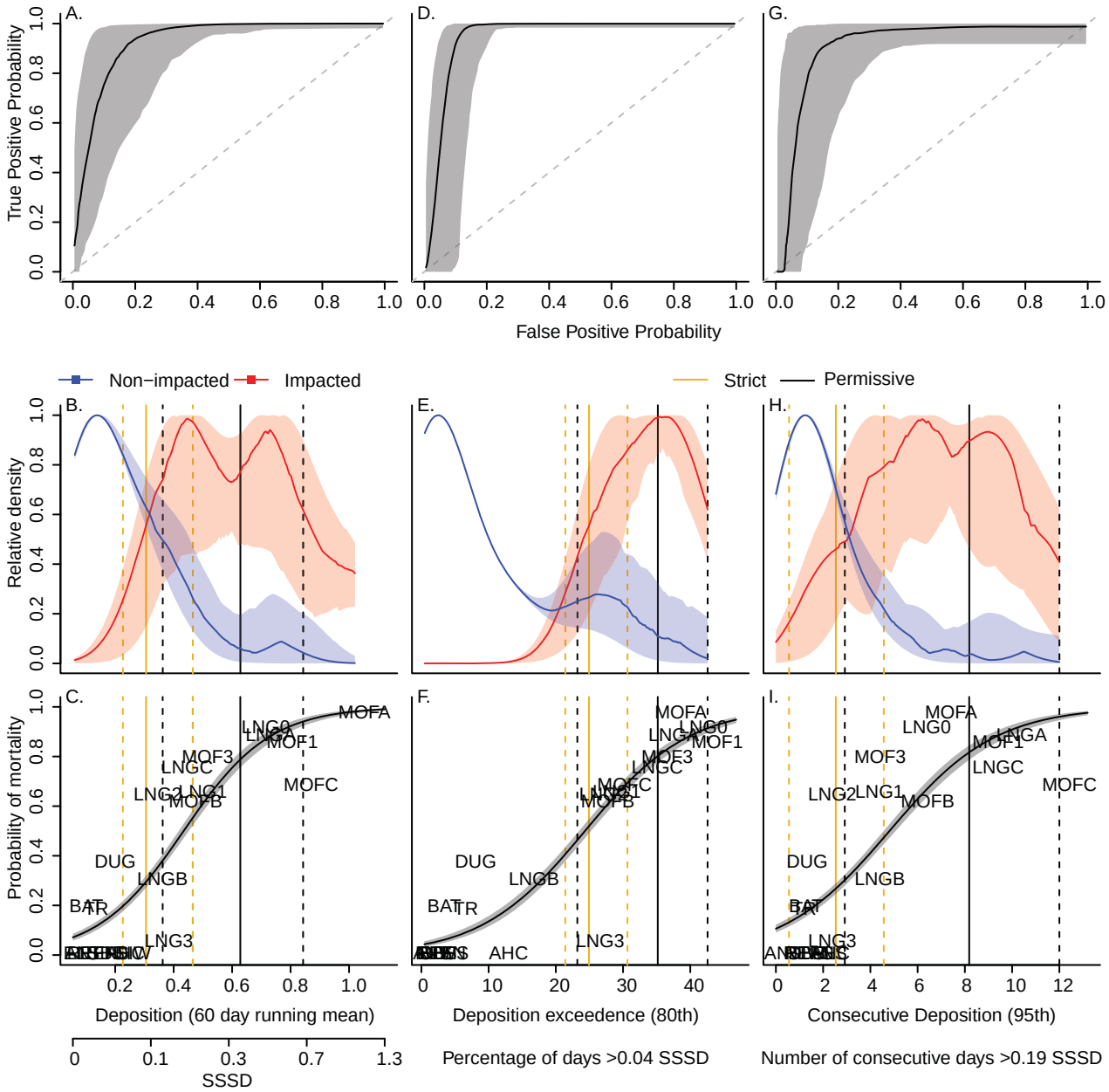

Fig. S6.1. Deposition based ROC curves and derived thresholds for running mean (panels A-C), percentage exceedance (panels D-F) and consecutive exceedance (panels G-I) exposure metrics. Upper panels show the estimated ROC curves (panels A, D and G) and middle panels the probability density functions for impact (red) and non-impact (blue) locations (panels B, E, and H). For both ROC curves and probability density plots, the solid curves represent the median values across 1000 iterations of random assignments of sites to non-impact and impact treatments (based on their probabilities of coral mortality), with shaded transparent areas showing the 95% confidence envelope. Lower panels (C, F and I) show the dose-response curves between each metric and the probability of coral mortality. Three separate thresholds were derived from the ROC curves: Permissive – representing the threshold required for a 5% false positive rate, black vertical line; Strict – representing the threshold that would provide a 95% true positive success rate, orange vertical line. Threshold 95% confidence bounds are represented by vertical dashed lines.

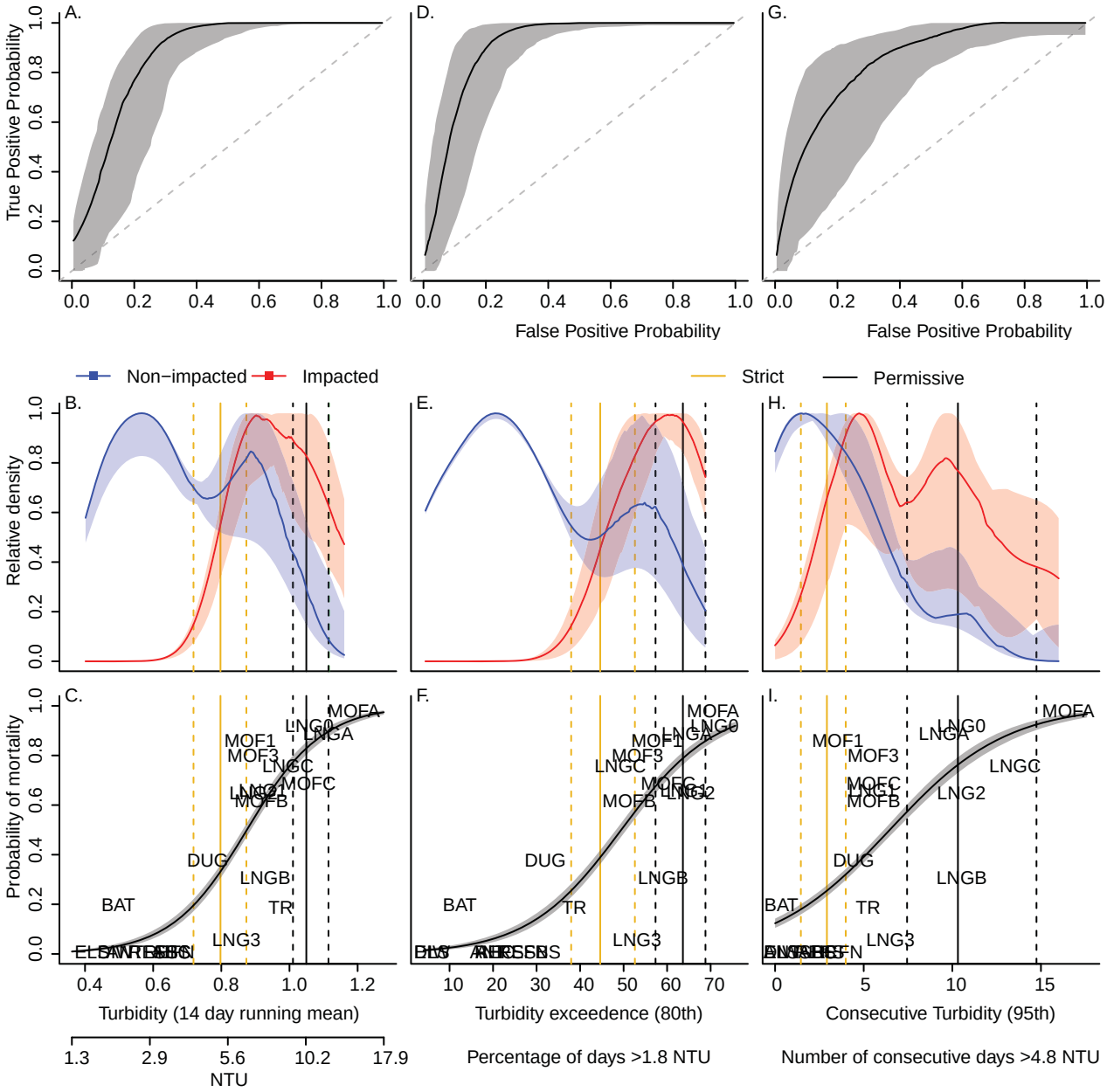

Fig. S6.2. Turbidity based ROC curves and derived thresholds for running mean (panels A-C), percentage exceedance (panels D-F) and consecutive exceedance (panels G-I) exposure metrics. Upper panels show the estimated ROC curves (panels A, D and G) and middle panels the probability density functions for impact (red) and non-impact (blue) locations (panels B, E, and H). For both ROC curves and probability density plots, the solid curves represent the median values across 1000 iterations of random assignments of sites to non-impact and impact treatments (based on their probabilities of coral mortality), with shaded transparent areas showing the 95% confidence envelope. Lower panels (C, F and I) show the dose-response curves between each metric and the probability of coral mortality. Three separate thresholds were derived from the ROC curves: Permissive – representing the threshold required for a 5% false positive rate, black vertical line; Strict – representing the threshold that would provide a 95% true positive success rate, orange vertical line. Threshold 95% confidence bounds are represented by vertical dashed lines.

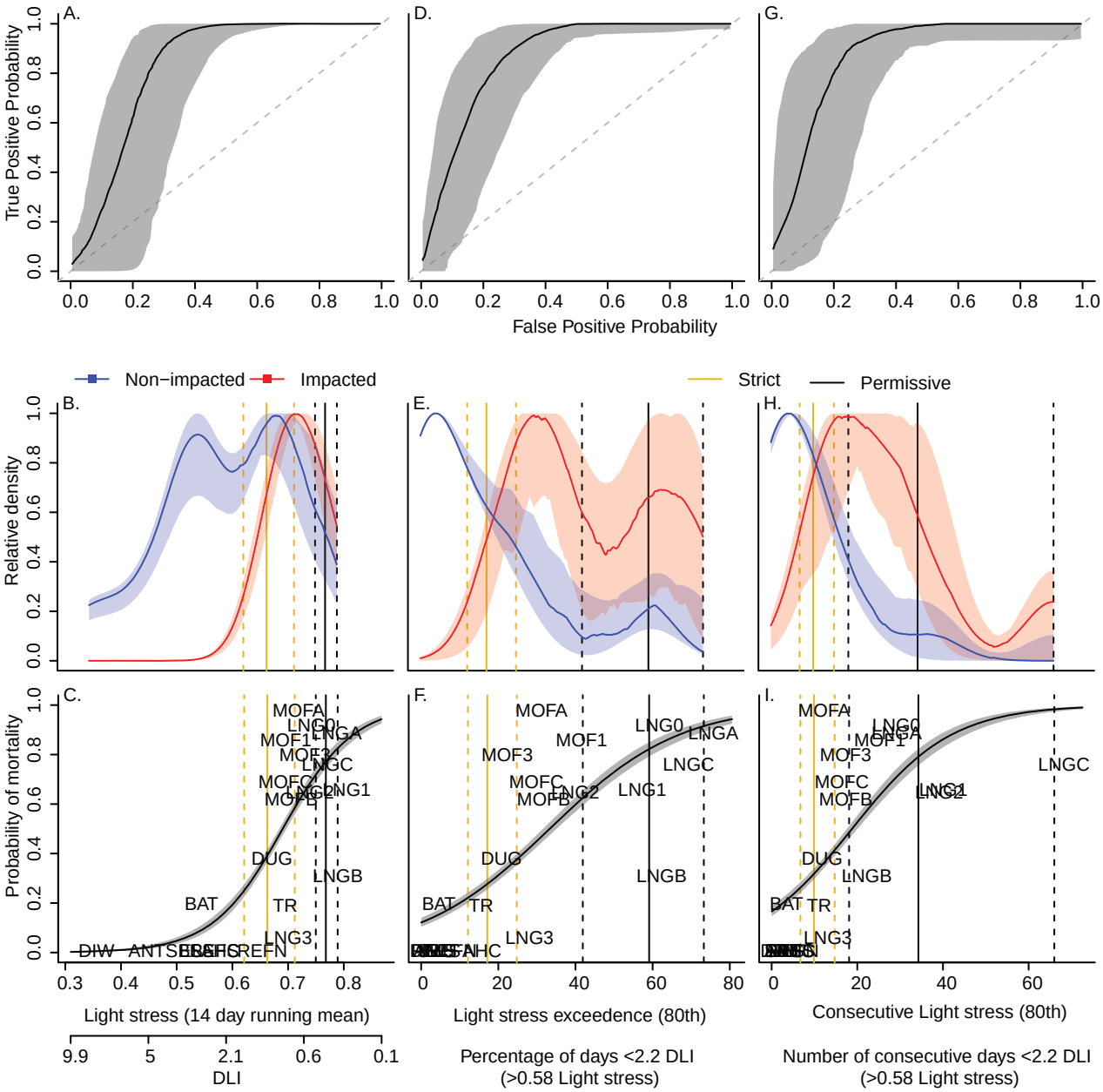

Fig. S6.3. Light stress based ROC curves and derived thresholds for running mean (panels A-C), percentage exceedance (panels D-F) and consecutive exceedance (panels G-I) exposure metrics. Upper panels show the estimated ROC curves (panels A, D and G) and middle panels the probability density functions for impact (red) and non-impact (blue) locations (panels B, E, and H). For both ROC curves and probability density plots, the solid curves represent the median values across 1000 iterations of random assignments of sites to non-impact and impact treatments (based on their probabilities of coral mortality), with shaded transparent areas showing the 95% confidence envelope. Lower panels (C, F and I) show the dose-response curves between each metric and the probability of coral mortality. Three separate thresholds were derived from the ROC curves: Permissive – representing the threshold required for a 5% false positive rate, black vertical line; Strict – representing the threshold that would provide a 95% true positive success rate, orange vertical line. Threshold 95% confidence bounds are represented by vertical dashed lines.

Project 4.9.4. Fisher R, Bessell-Browne P, Jones R (2019) Deriving and operationalizing thresholds for managing dredging impacts on coral reefs.

Deriving and operationalizing thresholds for managing dredging impacts on coral reefs

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Contents

1 Introduction

Dredging activities such as excavation and dredge material placement create turbid plumes of suspended sediment that can drift onto nearby sensitive habitats such as coral reefs. The suspended sediment can have a range of effects on marine communities (Erftemeijer & Lewis 2006, Bell et al. 2015, Wenger et al. 2017). For corals, suspended sediments can impact on their heterotrophic feeding ability, attenuate light thereby reducing rates of algal symbiont photosynthesis, and fall out of suspension causing smothering of both adults (Jones et al. 2016) and juveniles (Jones et al. 2015b).

There has been 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. This can be achieved by thresholds or guideline values that relate the physical pressures[1] to the biological response(s) and define exposure conditions above which effects will occur. For the case of dredging related impacts on corals, deriving threshold values can be complex, because there are many interlinked cause-effect pathways, acting alone or in combination (see also Jones et al. (2016)). As such, the most relevant parameter(s) may change according to dredging activities, sea-state, distance from the dredge etc, and it is difficult to identify which is the most relevant or important pressure parameter(s) at any given time.

The task of deriving guideline values is further complicated by the fact that corals exhibit flexibility in their heterotrophic versus autotrophic feeding (Anthony & Fabricius 2000) and many corals have lipid stores that can be used to maintain vital life processes allowing them to tolerate periodically poor water quality conditions (such as high SSCs and reduced light), providing these are interspersed with periods of favourable conditions. Furthermore, corals can differ markedly in their response to the various dredging related stressors, for example branching corals show greater sensitivity to light reduction and some massive coral are more sensitive to sediment deposition (Bessell-Browne 2017). Corals also exhibit substantial phenotypic plasticity, with individuals of the same species showing differing capacities to adapt to and/or tolerate poor water quality conditions (Rocker et al. 2017). It seems likely that benthic communities from clear versus turbid water environments may show substantially different tolerance thresholds to many dredging related stressors.

Guidelines can be developed from both *ex situ* laboratory-based (i.e. aquarium) studies and/or from *in situ* field-based studies. The laboratory-based studies outlined in Project 4.6 described a series of manipulative experiments designed to isolate and separate (disentangle) the variables, testing the effects of sediment or light attenuation or sediment deposition on corals alone, and sometimes in combination. Dose-response relationships and associated thresholds were also derived for some variables for a suite of coral species and morphologies (Bessell-Browne et al. 2017d).

It is also possible to develop guidelines from field studies where simultaneous measurements have been made of coral health and associated water quality. Some of the advantages of deriving guidelines from field studies are that they are less artificial, hence ecologically relevant; however, there are also some limitations. Firstly, it is impossible to repeat a study under identical conditions, thus replication is an issue. As dredging impacts on water quality can be extensive (>15 km, (Evans et al. 2012, Fisher et al. 2015)), and natural mortality can be highly variable so the scale of sampling required is generally prohibitively expensive. It is also very difficult to remove other confounding effects, such as natural cross shelf gradients in turbidity (Abdul Wahab et al. 2017). Secondly, field studies can be difficult to interpret because of the combination of pressures operating in unison (Jones et al. 2016). Lastly, field studies are also susceptible to uncontrollable external anomalies (Abdul Wahab et al. 2017), and these chance events (such as cyclones or marine heatwaves) can further confound interpretations.

[1] A *pressure* is a physical, chemical or biological change that has the potential to cause environmental change. In terms of dredging activities the term is used to signify elevated SSCs, sediment deposition, or reduced light availability from increased turbidity.

Monitoring studies generating datasets comprehensive enough to allow field–based derivation of guideline values are rare. However, one such project is the water quality and coral health data collected during the capital dredging project at Barrow Island[2], located 50 km off the Pilbara region of NW Western Australia (NW WA). This remarkable dataset has information on turbidity (as nephelometric turbidity units, NTU), photosynthetically active radiation (PAR), a relative index of sediment deposition (Ridd et al. 2001), as well as simultaneous observations of >1500 of individual corals, repeatedly photographed at approximately 2 weekly intervals throughout the dredging period. The combined water quality and coral health monitoring was conducted at multiple (>25) sites located along a clear gradient of water quality from 200 m to up to 30 km away from the dredging. Water quality guidelines were derived from the first 203[3] days of this data set for a range of pressure metrics, using novel statistical methods and a formal decision science approach (Fisher et al. 2018, Project 4.9.3). These thresholds were based on the probability that there was non-zero coral mortality at each site, accounting for environmental uncertainty and natural mortality at control locations – but were also framed within the context of probabilities of generating false alarms or failing to detect environmental impacts (i.e. Type 1 and Type 2 errors). The terms *permissive* threshold (with only a 5% probability of yielding false alarms), and *strict* threshold (with only a 5% probability of failing to detect significant environmental impacts) were coined by Fisher et al. (2018, Project 4.9.3). From a theoretical perspective, these approaches are appropriate because of the capacity to formally account for the uncertainty associated with the (always) imperfect ability for metrics of environmental stress to predict environmental impact. From a practical perspective, the exceedance of the *strict* threshold indicates that water quality conditions have reached levels above which mortality of corals is *possible*, but are not certain. Exceedance of the less conservative *permissive* threshold indicates that water quality conditions have now reached levels where negative impacts on corals are becoming *probable*.

While the thresholds derived by Fisher et al. (2018) are useful, there are a range of other complexities that could not be thoroughly addressed and/or reported. At a fine scale of analyses there are many issues and practical considerations for the implementation of derived guideline values, and there is considerable uncertainty in how to treat water quality data used to derive and implement thresholds. For example, while most dredging impact modelling exercises focus on suspended sediment concentrations (SSCs) of dredge plumes, in reality turbidity and/or loss of light are more easily measured in the field. In addition, both turbidity (NTU) and daily light integral (DLI, mol quanta m^{-2} d^{-1}) can be highly skewed, making statistical treatment of these variables challenging (Jones et al. 2015a). How to treat data collected during cyclones and other severe weather events is another question frequently posed (see Fox (2016)), and the implications, if any, for including or excluding data influenced by extremes is unknown.

At a higher level there are a range of aspects of threshold derivation that warrant further discussion, given the complexities associated with measuring and understanding dredging related pressures, as well as the variable responses and sensitivities of different corals and coral communities. There are a bewildering array of ways that water quality time series can be summarised into dredging related pressure metrics and used to derive guideline and thresholds values (e.g. see Fisher et al. (2018)). While it is clear there are trade-offs between intensity, duration and frequency of dredging pressures and associated impacts on coral health, in practice temporal scales of thresholds are not explicitly articulated (for example Foster et al. (2010)) and cumulative dredging related pressure (capturing both intensity and duration of impacts) is often not accounted for. Alternative methods of derivation yield different threshold values (Fisher et al. 2018, Project 4.9.3) and these need to be understood in an operational context. Furthermore, field and/or experiment based methods of threshold derivation focused on coral tolerance limits are not always possible, so an understanding of how these compare to other strategies for deriving thresholds using only baseline water quality conditions (such as the P_{50} P_{80} of the ANZECC/ARMCANZ guidelines (ANZECC/ARMCANZ 2001)) would provide guidance on operalisation of thresholds across broader spatial scales.

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[2] Barrow Island: WA Environmental Protection Authority Bulletin 1221 Ministerial Statement No. 800

[3] While the dredging lasted for 530 d, the period after the first 203 days was impacted by a bleaching event

Finally, the issue of sediment deposition remains a challenging problem in the management of dredging related impacts. Smothering of corals by deposited sediment represents a direct mortality pathway (Bessell-Browne 2017), but also causes sub-lethal stress potentially decreasing coral fitness (Bessell-Browne et al. 2017c, Duckworth et al. 2017) and inhibiting coral settlement (Ricardo et al. 2017). Real time measurements of deposition remain a challenge, and simple strategies for estimating likely sediment deposition loads and the extent of sediment deposition zones around dredging activities need to be further explored.

In this report we explore some of the complexities associated with deriving and operationalising threshold guidelines for corals, provide context for some of the many assumptions underlying the threshold derivation in Fisher et al. (2018), and the logic behind the theoretical approaches used. We explore some practical considerations for threshold implementation (section 2) and issues that are important in threshold derivation (section 3). We then explore the relationships between sediment deposition (section 4) and light attenuation (section 5) and coral health. We finish with a discussion around threshold development for impact prediction and management (section 6), including thresholds for delineating spatial zones of impact and triggering management actions.

2 Practical considerations for threshold implementation

2.1 Capital dredging projects in NW WA and water quality measurements

The analyses below have been made possible by the availability of water quality and in some cases coral health datasets collected during four large scaled dredging projects in the Pilbara region of NW WA. Water quality data was collected at 32 sites for the Burrup Peninsula Project, 26 sites during the Barrow Island project (of which 22 were used in analyses, see Fisher et al. (2018)), 15 sites during the Cape Lambert project and 22 sites from the Onslow (Wheatstone) project (Figure 1). All projects had sites spanning distances of up to ~30 km from the location of dredging activities. While the projects were relatively near to each other, spanning a total distance of <250 km, they did occur in different marine settings and therefore represent a range of coral reef environments. The Burrup Peninsula project was conducted in an enclosed inshore turbid reef environment (Mermaid Sound in the Dampier Archipelago), the Cape Lambert project occurred in an exposed nearshore cape or headland, the Barrow Island project occurred in an offshore 'clear water' environment, and the Wheatstone project involved dredging primarily in the nearshore (for a coastal facility) and along a 25 km channel with a distinct inshore to offshore gradient in water quality (Figure 1). Full details for each site sampled, including total baseline and dredge period sampling days, water depth (where available) and distances of the monitoring sites from the main dredging activities are listed in (Fisher et al. 2015, Jones et al. 2015a, Abdul Wahab et al. 2017). All water quality data was processed similarly to ensure data integrity and to remove potentially erroneous values. Full details of the data processing can be found in Jones et al. (2015a).

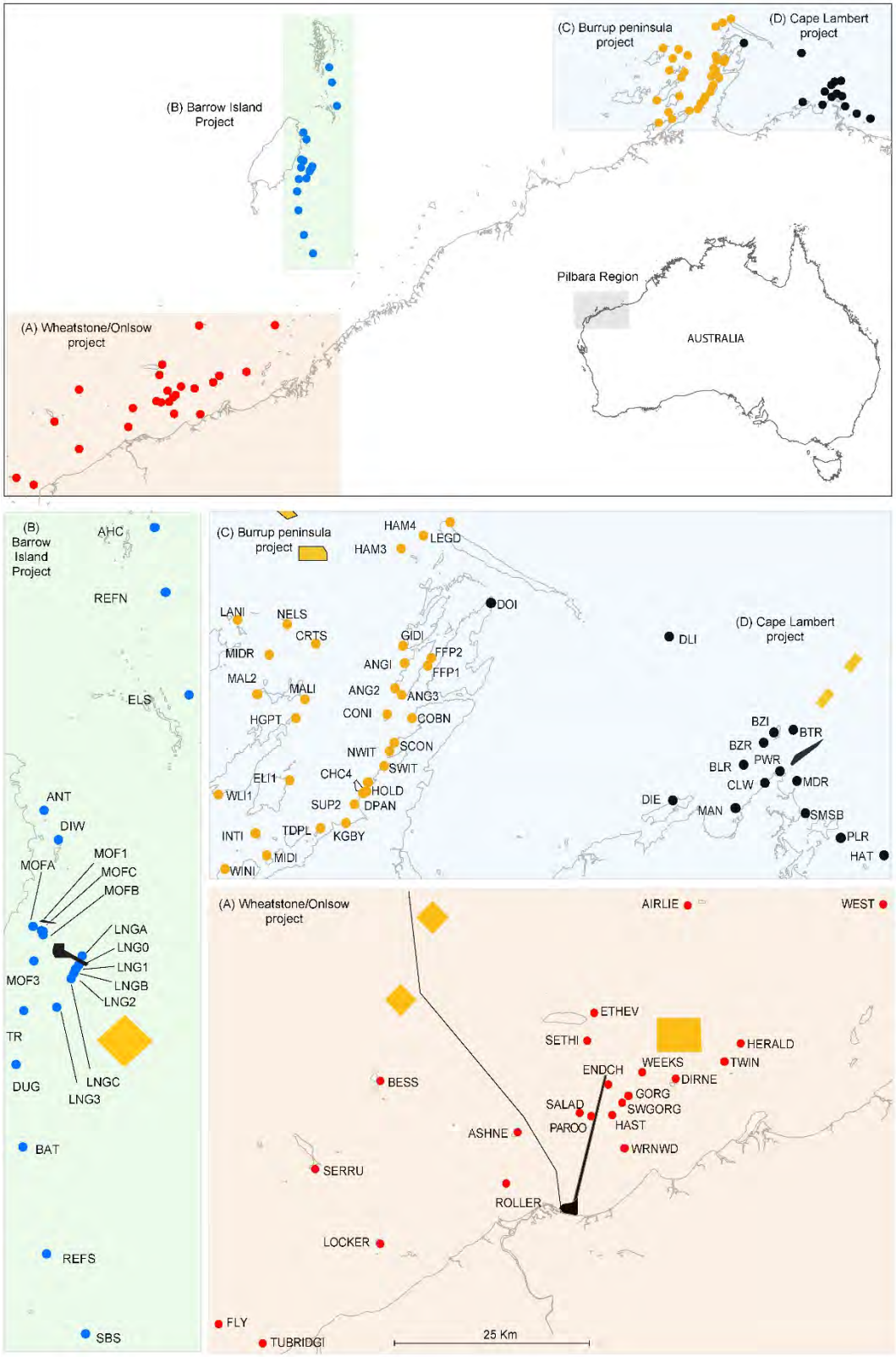

Figure 1. Water quality monitoring sites for four capital dredging projects in the Pilbara region (Western Australia) since 2009, Including the Onslow (Wheatstone) project (WA Environmental Protection Authority Bulletin 1404 Ministerial Statement No. 873) (A, black symbols); the Barrow Island project (Gorgon Gas Development Barrow Island Nature Reserve: WA Environmental Protection Authority Bulletin 1221 Ministerial Statement No. 800)(B, blue symbols), Burrup Peninsula project (WA Environmental Protection Authority Bulletin 1259, Ministerial Statement No. 757) (C, yellow symbols), and Cape Lambert B project (WA Environmental Protection Authority Bulletin 1357, Ministerial Statement 840) (D, black symbols). Ministerial approval statements are searchable on the WA EPA website.

Water quality monitoring programs in recent capital dredging projects in Western Australia have typically focused on measuring turbidity, using optical back scatter technologies (nephelometers), which can be used to provide proxy estimates of SSCs. Focusing on SSCs and using nephelometers for adaptive management of dredging projects was first introduced in the inshore GBR as early as 20 years ago (Benson et al. 1994, Koskela et al. 2002, Orpin et al. 2004). The emphasis on SSCs is intuitive because it is associated with the sediment itself which is released into the water column by dredging, but importantly also because SSCs can be modelled during the pre-dredging EIA stage using coupled sediment transport and hydrodynamic models that predict sediment transport and fate. The many laboratory-based experiments involving exposure of corals and other benthic organism (see Jones et al. (2016)) have also related physiological effects to gravimetrically determined suspended sediments, and so SSC (as mg L^{-1}) is the a common term for pressure field monitoring and impact prediction.

Nephelometers provide a good proxy or surrogate measure of SSCs (i.e. nephelometrically-derived SSCs), and thresholds based on NTU have been derived using the field datasets at Barrow Island from the Gorgon dredging campaign (Fisher et al. 2018). However, for NTU based thresholds to be used in EIA and dredge management plans, conversion factors are needed to relate NTU to SSC and these can introduce uncertainty and error. Conducting NTU–SSC conversions is not trivial and the conversion factors can vary depending on how they are calculated (MScience 2009). There can be pronounced differences in the NTU–SSC relationships across time and space (Fox 2016), depending on the sediment type and instrument geometry. Although most conversion factors are typically linear (although see Table 1) the relationship may be curvilinear over the 4 orders of magnitude range of NTU values sometimes encountered even naturally (Orpin et al. 2004). Macdonald et al. (2013) suggested a linear relationship ranging from 1 and 4 (see also Orpin et al. (2004)). While NTU–SSC conversions were carried out for all of the capital dredging programs studied (Table 1), these were spatially and temporal sporadic, with only a few relationships developed through time or across sites within each location. During the Barrow Island dredging program, which was used to derive the guidelines values for corals in Fisher et al. (2018), the SSC–NTU conversions relationships ranged from 0.8 to 3.2 (Table 1), highlighting that considerable error can be involved in converting NTU based thresholds into values of SSC. Using a mean conversion factor across the sites at Barrow Island of 1.8×, the *strict* (possible effects) and *permissive* (probable effects) running 14 day geometric mean NTU thresholds of 5.3 and 10.2 NTU would represent SSC values of 9.6 and 18.5 mg L^{-1}.

Table 1. NTU-TSS conversions reported and used across the four dredging programs showin in Figure 1.

Site/Conversion		reference
Barrow Island project: WA Environmental Protection Authority Bulletin 1221 Ministerial Statement No. 800		
BIG =1.9×NTU, DUG =2.08×NTU, LONE =2.31×NTU, LOW =1.58×NTU, LNG0 =0.928×NTU, LNG1 =2.72×NTU, LNG2=1.86×NTU, LNG3 =2.3×NTU, MOF1 =1.22×NTU, MOF2 =0.8×NTU, MOF3 =1.94×NTU, SBS =1.62×NTU, AHC =3.26×NTU, ANT =1.5×NTU, BAT =1.79×NTU		(Chevron 2012)
Burrup Peninsula project: WA Environmental Protection Authority Bulletin 1259, Ministerial Statement No. 757		
(CHC4, HOLD & DPAN)	SSC=1.47×NTU (calculated as a mean of daily conversion factor) SSC=1.174×NTU (calculated by linear regression)	(MScience 2009)
Cape Lambert B project: WA Environmental Protection Authority Bulletin 1357, Ministerial Statement 840		
TSS = A×NTU*exp(B×NTU)d + C, Where A = 0.67035933, B = 0.25565056, C = 0.27502312 and d = 0.0391		(In-situ Marine Optics 2011)
Onslow: WA Environmental Protection Authority Bulletin 1404 Ministerial Statement No. 873		
TSS=1.1×NTU (assumed correlation)		Reference not provided

In conclusion, proxy estimates from nephelometers (i.e. nephelometrically-derived SSCs) are currently the only available technique for routine monitoring of elevated SSCs, and given the variability in NTU–SSC conversion ratios, if guidelines are based on SSC (as opposed to light), care must be taken to check and reconfirm the conversion factors at frequent intervals throughout during dredging programs.

2.2 Data aggregation

Seawater-quality data are usually recorded at relatively fine temporal scales (e.g. minutes), and aggregated to coarser time scales for the purposes of reporting. The summary statistics used (mean versus median etc), as well as the temporal scale adopted (hours, days, weeks) can dramatically affect the interpretation of the data (Jones et al. 2015a). Both NTU and the daily light integral (DLI, mol quanta m^{-2} d^{-1}) can be highly skewed making statistical treatment of these variables challenging (Jones et al. 2015a). Short periods of high SSCs or low light are ecologically significant and the importance of these events are not clear or reflected in median values and especially over longer term averages (Gaines & Denny 1993). If the hazards associated with dredging are to be characterised thoroughly, they need to be expressed both with respect to changes in central tendency, but also in terms of changes in upper (e.g., maximum, 95th percentile) and lower bounds.

In addition, NTU represents a positive stress metric (high values are expected to result in mortality), whereas DLI is negative (low values are expected to result in coral mortality). In developing the thresholds for dredging management (Fisher et al. 2018), developed stress indices for both NTU and PAR (as DLI) based on:

$$Turbidity\ stress = log10\ (NTU+1) \qquad (1)$$

Where: NTU is nephelometric turbidity unit.

A logarithmic form was selected as NTU readings are highly right-skewed and can cover several orders of magnitude (~0–>100) (Jones et al. 2015a). Using NTU on a log scale is also (perhaps arguably) appropriate ecologically, as a change from 1 to 10 is likely to have similar biological consequences as a change from 10 to 100 (except where this may indicate potential sedimentation, but see discussions of SSSD below).

To capture stress associated with reduced light levels, a light stress index was developed to represent the relative loss of light available from the increased turbidity associated with dredging. This was calculated as:

$$Light\ Stress\ Index = 1 - \left(\frac{DLI}{30}\right)^{\frac{1}{3}} \qquad (2)$$

Where: DLI is the daily light integral (mol quanta m^{-2} d^{-1});

30 was chosen to represent the maximum sub-surface DLI (actual maximum observed values were ~28 DLI at this location);

$\frac{\mathrm{DLI}}{30}$ was subtracted from 1 to ensure that stress is represented as increasing values of the Light stress index, such that high scores represent large amounts of lost light (i.e. high levels of stress are expected to lead to high levels of mortality); and

A cube-root transformation was used to remove severe right skew in relative DLI, ensuring greater spread among higher stress index values (extreme low light) and a relatively normal distribution.

While the derived stress indices have proven robust predictors of coral mortality, many people working with water quality data tend to work with raw units of NTU, calculating running means on their original scale (e.g. (Fisher et al. 2015, Jones et al. 2015a, Jones et al. 2016, Abdul Wahab et al. 2017)). Calculating running mean NTU on a log scale equates to calculating a geometric mean (Crawley 2005), and can yield lower mean values than a mean calculated on a linear scale. At very low turbidity levels (<1 NTU) the differences between the two calculated running means are very small (Figure 2A). However, as running mean NTU increases differences can

be over two fold, and highly unpredictable at mid to high turbidity levels (Figure 2A). Differences also tend to be greater for longer running mean times scales (Figure 2A).

Similarly, running means based on light stress and back transformed to units of DLI (mol quanta m^{-2} d^{-1}) can also yield lower values compared to a running mean based on untransformed DLI (Figure 2B). However, for DLI, these differences are small across a broad range of DLI values (2–30 mol quanta m^{-2}), only increasing for extremely low values (<0.2 mol quanta m^{-2}, Figure 2B). For very low DLI levels, differences can be substantial (>400%, Figure 2B) – although the error in units of mol quanta m^{-2} d^{-1} (rather than expressed as a percentage) is actually very small (i.e. a 200% error at 0.1 mol quanta m^{-2} d^{-1} is an error of ~0.2 mol quanta m^{-2} d^{-1}).

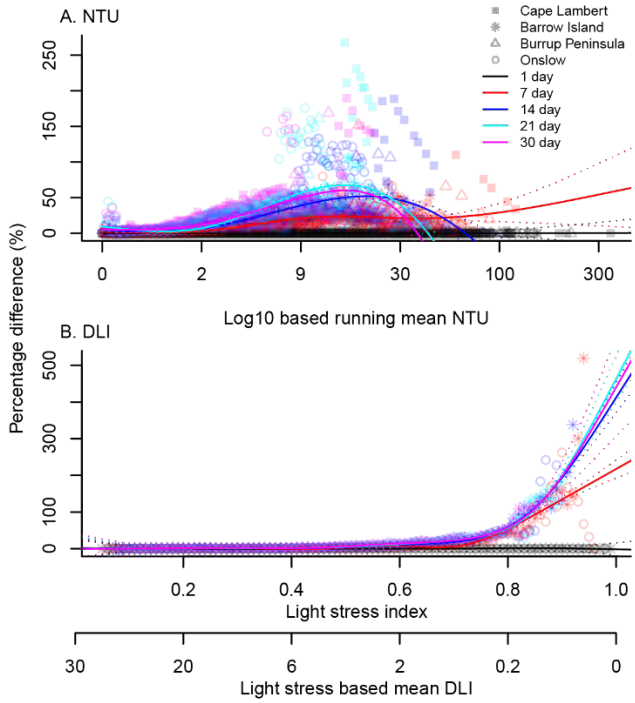

Figure 2. Percentage difference in calculated running mean values obtained using raw NTU (A) and DLI (B) compared to values obtained based on the respective stress indices (turbidity = log10 (NTU+1); light stress = 1-$(DLI/30)^{1/3}$). Percentage difference is plotted as a function of back-calculated stress index values, with symbols indicating the dredging program from which the data were sourced and colours indicating differences calculated for each of the running mean time scales (1, 7, 14, 21 and 30 days). Solid lines indicate generalised additive model fits (GAMS) with 95% confidence bands (dotted lines).

In conclusion, differences in the estimates obtained based on geometric versus absolute mean NTU highlights the potential impact that the type of statistical summaries can have on subsequent reported values. How data are handled, in the context of transformations and averaging, must be carefully considered and always clearly reported, particularly where running mean values are used as thresholds in management.

2.3 Water quality, cyclones and absolute versus relative guidelines

Managing local impacts in the face of multiple natural stressors across broad spatial and temporal scales (Hughes & Connell 1999) remains a considerable challenge in environmental science and not only for dredging activities. Extreme weather events such as tropical cyclones can significantly affect water quality monitoring programs associated with dredging projects, and many of the recent dredging programs in tropical NW Australia have been heavily influenced by tropical cyclones (Jones et al. 2015a, Abdul Wahab et al. 2017, Ricardo et al. 2017). How to treat data collected during cyclone and other severe weather events where extreme turbidity levels occur is not straightforward (Fox 2016).

Using the baseline water quality data from 4 capital dredging projects in NW WA (see Figure 1) the 80th percentile (P_{80}) of the 14 d running mean NTU data was compared with and without data collected during cyclones. For the analyses, all turbidity data collected over the baseline periods were excluded from the day before and the day after each system was upgraded to (or downgraded from) a cyclone (based on information from the Australian Bureau of Meteorology), with all cyclones occurring anywhere in the vicinity of the NW WA considered.

The P_{80} of the baseline turbidity was found to be as much as 60% higher if cyclones are included in the analyses (Figure 3). The large variation between the dredging projects stems in part from the fact that there were no

cyclone events during the baseline data for the Burrup Peninsula project and the Barrow Island dredging projects (Figure 3). However, water quality conditions during the baseline periods for the Wheatstone (Onslow) dredging program were markedly influenced by the cyclones (i.e. ~30% increase in running 14 day mean NTU on average across sites).

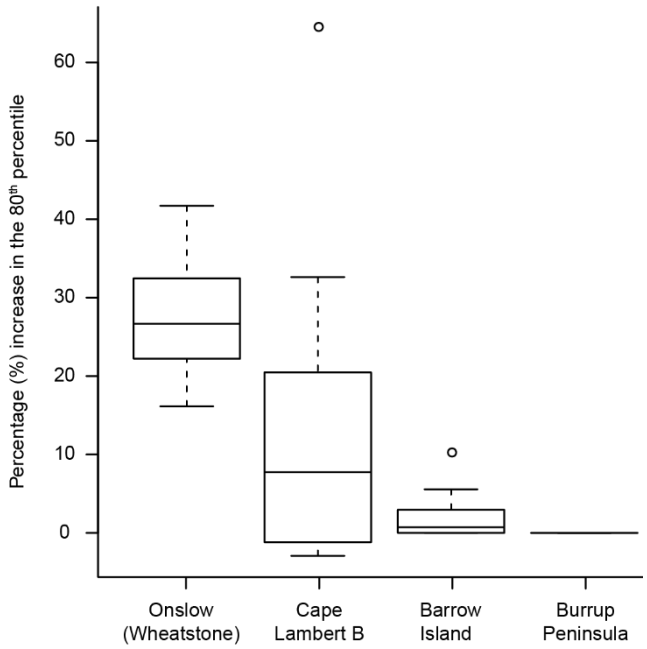

Figure 3. Percentage increase in estimates of the baseline (pre-dredging) 80th percentile NTU (for 14 d running mean) based on data including cyclones compared to data where cyclone periods are excluded. The boxplots show the variation across sites from four dredging projects in the Pilbara (Figure 1). The middle line of the boxes indicates the median, upper and lower hinges the interquartile range, with the whiskers extending to the most extreme data point no more than 1.5 times this interquartile range. For the analyses all turbidity data collected over the baseline periods were excluded for a few days before and after the system was upgraded or downgraded to a cyclone (based on information the Bureau of Meteorology). Excluded cyclones include: Nichols (2008), Billy (2008), Dominic (2009), Bianca (2011), Carlos (2011), Iggy (2012), Lua (2012), Mitchell (2012-13), Narelle (2013), Rusty (2013) and Christine (2014).

Including cyclones during baseline data time series for the purposes of characterising the background water quality conditions may be theoretically appropriate as these events do occur, and form part of the natural exposure regime experienced by benthic communities. However, from a more practical perspective, although the NW WA region is cyclone prone, probabilistically they are quite rare events and on average only two cyclones cross the entire coastline each year. Capturing extreme oceanographic and/or meteorological events with such low frequency would require collecting baseline data over extended (decadal) periods and that is simply not feasible –and yet including data collected during cyclones could highly distort percentiles. If the purpose of water quality monitoring is to alert dredging proponents to conditions where environmental damage may occur (so as not to breach permit conditions which may have significant consequences) it seems reasonable that cyclone affected water quality monitoring data should be omitted from baseline data for the purposes of threshold derivation. This will allow a better assessment of the long-term natural background conditions without the statistical distortions associated with including extreme events.

In practice, removing cyclone-influenced data collected during a baseline period is more difficult than it seems. Cyclone intensity, size, duration and proximity vary substantially, influencing the wave-climate (Puotinen et al. 2016) and natural levels of seabed resuspension. Small but less intense cyclones (or even tropical lows) passing nearby may have more significant effects on turbidity and light availability than larger, more intense, but distantly located cyclones. Future work should attempt to understand the link between cyclone generated wave conditions and the water quality hazard profile, and how cyclone events should be managed during dredging projects. It would be of value to derive a set of threshold wave heights that result in significant alterations of water quality conditions, and to explore how such thresholds vary spatially. Furthermore, it would be of considerable interest to investigate if (and how) dredging alters the broader sediment regime of a location, potentially leading to changes in wave height thresholds that result in elevated turbidity. Such changes, if they persist over the long term, should be considered in risk assessments of proposed dredging activities.

Whether or not to exclude cyclone influenced data during the dredging period for the purpose of assessing threshold values is more difficult, and is synonymous to the suggestion that guideline values should be assessed relative to reference sites (see Figure 2 in Erftemeijer et al. (2012), and more quantitatively in Table 3 of Foster et al. (2010)). Such suggestions arise from the argument that the elevated turbidity and reduced light availability caused by cyclones (or in fact any natural turbidity event), are not caused by dredging, and should therefore have no consequence for dredging water quality management. However, this logic seems contrary to the purpose of guidelines – to alert dredging proponents of possible effects – as benthic communities cannot discriminate between natural and dredging related turbidity events. Similarly for benthic light availability, periods of increased storminess and high cloud cover (or seasonal cycles in light availability associated with solar declination) will reduce benthic light levels. Natural disturbance regimes involving elevated SSCs and low light are inherent features of coral reef ecosystems, and this has to be considered when using guidelines, as equivalent natural or anthropogenic changes confer the same metabolic and physiological costs to corals. Implicit in the development of water quality thresholds in Fisher et al. (2018) and below, is a concept of absolute and not relative (to reference locations) guideline values.

In conclusion, water quality data (turbidity, PAR and sediment deposition) affected by cyclones in the baseline phase needs to be examined on a case-by-case basis and possibly quarantined if baseline data is used for deriving thresholds. The size and proximity of the cyclone needs to be considered, but also whether the cyclones have had 'extreme' effects on the data. Water quality data (turbidity, PAR and sediment deposition) that is affected by cyclones during the dredging periods should be included in monitoring programs. Similarly, absolute guidelines (i.e. x mg L^{-1} for y days or over period y) should be used for guidelines as opposed to relative (to reference locations) guidelines (i.e. x mg L^{-1} for y days or over a period of y days, over a background of z mg L^{-1}). Where natural events result in threshold exceedances for a given impact zone (see below), it may be permissible to continue operations if there is confidence that the activities are not significantly contributing to the water quality conditions leading to exceedance.

3 Deriving thresholds

3.1 Threshold durations

Impacts of dredging on water quality are highly ephemeral (Jones et al. 2015a) and spatially complex (Fisher et al. 2015), with changes to water quality occurring across both acute (changes to the extremes) and chronic scales (changes to long term conditions). Because there are trade-offs between intensity, duration and frequency of dredging pressures and associated impacts on coral health, it is important that guideline values incorporate a temporal component. However temporal scales of thresholds are not always clearly evident, nor is the relationship between duration and intensity explicitly articulated. For example, a guideline value of SSCs >10 mg L^{-1} for less than 20% of the time has been associated with moderate impact to corals in Singapore (Foster et al. 2010); however, such guideline values are non-specific as to when the exceedances occur (i.e. to the overall loading). For example 20 d in a row during a 100 d dredging period (i.e. 20%) may have more significance for corals than 20 d interspersed among a 100 d period. Similarly, 20 d of a 100 d dredging period (i.e. 20%) may have substantially greater impact to benthos than 10 d of a 50 d dredging period, or 5 d of a 20 d dredging period (both also 20%), because 20 days of exposure represent a potentially greater cumulative impact of exposure to sediment stress. In addition, an exceedance of >110 mg L^{-1} (a substantial exceedance) compared to ~15 mg L^{-1} (a marginal exceedance) should not be treated similarly, as from a physiological perspective, there will be different consequences for benthic communities which needs to be accounted for to allow dredging proponents to fully understand the total pressure exerted on a community over time. Fisher et al. (2018) explored the use of running means as suitable pressure metric for guideline values, combing elements of both intensity and duration, where periods of very poor water quality are incorporated into an unweighted running mean of the sampling period in absolute terms.

While it is clear that duration must be captured in threshold development, one of the key questions is how long should the period of threshold calculation be – days, weeks or months? Addressing this issue requires an

understanding, amongst other things, of the speed of the biological responses. The laboratory based studies in Project 4.6 found that in response to very high SSCs (and the associated light reduction), or in response to light reductions alone, corals can undergo photo-adaptation in a matter of days by increasing the pigment concentrations of the algal symbionts. Photo-adaptive changes in the density of the symbiotic algae were also noted over a period of 7–20 days. Under extreme light deprivation (i.e. darkness or near darkness) loss of algal symbionts and bleaching occurred in ~1 week and partial mortality occurred in >10 days. Thus, from a physiological perspective, water quality guidelines framed over time periods of 1–2 weeks seem most appropriate as they align with sub-lethal and lethal responses of corals. A two week time frame for coral specific thresholds is further supported by (Fisher et al. 2018), who calculated running means across 1–60 day time scales for the Barrow Island data set and found that a 14-day running mean showed the strongest relationships with the probability of non-zero coral mortality for both turbidity and light stress (although a 60 day running mean was most appropriate for deposition).

From a practical, operational, perspective one to two weeks is a useful time-frame for water quality guidelines as periods of a few weeks allow dredging proponents time to plan and enact adaptive management activities if required (such as moving dredges, changing dredging modes or production rates etc). Also, having a comparatively short time component associated with a threshold (i.e. a running mean over weeks as opposed to months), should mean that if all dredging activities were to cease, the running mean values would respond (i.e. improve) relatively quickly, which would not occur if averaged over months. Other operational advantages of a 1–2 week running mean period for guidelines are associated with the influence of natural events such as storms or tidal cycles on water quality. The Australian Bureau of Meteorology has a strong and reliable weather forecasting capability out to around 7 d (Next Generation Forecast and Warning System (Munro & Munro 2011)), and natural turbidity generating events such as storms or weather fronts are thus more foreseeable. Similarly, the water quality analysis of baseline water quality conditions during several dredging program in WA (see Project 4.2.1, Appendix 1), identified distinct cycles (periodicity) in turbidity and light availability associated with spring-neap cycle centred around 14 d. If water quality monitoring during dredging suggests the approach of a guideline value requiring intervention, then reliable information of future weather patterns or a future spring tide sequence can be factored into the proponent's decision making process.

Maintenance dredging projects in Western Australia are typically 4–10 weeks, slightly longer than in Queensland where the average maintenance dredging duration is generally 2–4 weeks (Ports Australia 2014b). The duration of capital dredging projects in Australia has ranged from a few weeks to up to 2.5 years, although most extend for 4–6 months. Even within extended capital dredging projects, dredging often occurs across multiple locations (for example along a channel) and dredging pressures may occur on individual areas for a much shorter time. Thus basing thresholds on shorter time periods (1–2 weeks) is likely to be functionally useful for both shorter (maintenance) and longer (capital) dredging campaigns.

In conclusion, time components need to be built into guideline values that explicitly allow the calculation of accumulated pressure encompassing intensity and duration. There are multiple lines of evidence from physiological laboratory and field based studies that a 2-week (14 d running mean) period is an appropriate guideline time scale (at least for suspended sediments and associated light reduction). This is also a useful time frame from an operational and logistical perspective. For impact prediction at the EIA stage modelling scenarios over a 2-week period is suitable, although other longer periods may be more suitable and a range of metrics capturing different exposure profiles should be explored (discussed further below).

3.2 Use of baseline data for threshold development

Guideline values can be derived using a range of methods, and how they are derived depends both on the data available, as well as the intended use of the threshold. Where data are available on the sensitivity of benthic taxa of interest (such as corals) across a gradient of exposures (either in the laboratory or the field), thresholds can be based on dose-response curves, either explicitly accounting for probabilistic errors (see Fisher et al. (2018), or using the EC_{10} and EC_{50} approaches typically adopted in toxicology (e.g. Bessell-Browne et al. (2017d)). When

no data are available on the sensitivity of benthic organisms (to sediments or to altered environmental conditions), one approach is to base thresholds statistically on upper percentiles of information collected during the baseline phase. The rationale behind this relies on the fact that stressors (such as turbidity) are naturally present in the environment and that the existence and persistence of the benthic community is evidence of resilience to the natural exposure levels, and that the community will be protected as long as the system is managed such that impacts do not exceed some high-order percentile of these natural levels (ANZECC/ARMCANZ 2001, McArthur et al. 2002). One such strategy would be to use the P_{50}–P_{80} approach of the ANZECC/ARMCANZ guidelines (ANZECC/ARMCANZ 2001). In this method a trigger may be considered exceeded when the median value (P_{50}) for the impact location exceeds the P_{80} of the baseline state. The basis of this (i.e. P_{50} versus P_{80} as opposed to P_{50} versus P_{90} or P_{95} etc.) is somewhat arbitrary, but also pragmatic and associated with the notion of the developers that such a change (i.e. P_{50}–P_{80}) represents a measurable enough perturbation that is worth investigating (Fox 2001).

The rich water quality and coral health dataset collected during the Barrow Island dredging project provides a unique opportunity to compare different methods for deriving thresholds. The thresholds of Fisher et al. (2018) were based on probabilities of non-zero partial mortality (relative to mortality observed at control sites) [4] and were derived for a range of dredging related pressure metrics. The thresholds are applicable when the goal is to prevent any elevated mortality associated with dredging, as would be the case where regulations stipulate that water quality may be impacted by dredging but should not result in coral mortality (i.e. for an identified zone of influence (EPA 2016)[5]). Fisher et al. (2018) derived two distinct sets of thresholds that delineate between the *strict* thresholds[6], which focus on maximizing statistical power to detect potential impact (at the possible expense of higher false alarm rates) and indicate that mortality of corals is *possible*; and *permissive* thresholds[7], which focus on minimising false alarms (Fisher et al. 2018) and indicate that mortality of corals is *probable*. Estimated threshold values for 14 day running mean turbidity and light stress for these two types of thresholds differ substantially, highlighting the fact the depending on the specific goal of the threshold, different values are required (Figure 3).

The ANZECC/ARMCANZ P_{50}–P_{80} approach was used with the Barrow Island water quality data set and compared to the thresholds derived by Fisher et al. (2018) (see Figure 4). The analysis shows that the P_{50}–P_{80} approach is far more conservative than even the *strict* (possible effect) thresholds of Fisher et al. (2018), which in turn is based on highly conservative estimates of the probability of non-zero mortality of corals. Furthermore, it is only the upper range of the P_{95} boxplots that overlap, even with the most conservative threshold, suggesting another approach, the 95th percentile approach adopted by McArthur et al. (2002), may also be too conservative, at least at Barrow Island (Figure 3).

The baseline time series at many of these locations are relatively long (>1–2 years in most cases), and the percentiles probably captured the true range of background water quality conditions of sites at this clear-water location. While the percentile estimates in Figure 4 are for water quality data sets where cyclones events were removed, the same analysis including data collected during cyclones (during the baseline period) did little to shift the percentile values closer to the estimated tolerance thresholds (data not shown). This is largely because of the relatively small impact of cyclones on water quality conditions at Barrow Island, which is not necessarily be the case for other dredging projects (see Figure 3).

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[4] These 'probabilities of non-zero mortality' in effect represent a specific formulation of the uncertainty bounds associated with estimated no effect concentrations (NEC), the mean of which would be represented by a 50% probability.

[5] Note that while the approach could be theoretically modified to establish thresholds based on probabilities of mortality effects greater than zero (e.g. a 30% relative decline, as might be applicable to a Zone of Moderate Impact) this could not be done at Barrow Island because mortality was insufficient (<30%) even at very high impact locations

[6] Strict = more conservative = possible effects - with only a 5% probability of failing to detect significant environmental impacts

[7] Permissive = less conservative = probable effects - with only a 5% probability of yielding false alarms.

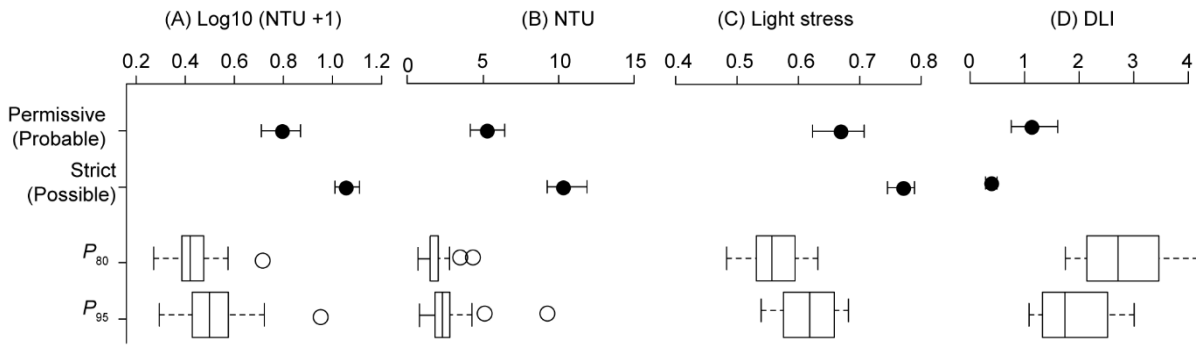


Figure 4. Comparison of 14 day running mean NTU and light thresholds estimated using data from the Barrow Island project. Solids circles with error bars show threshold values based on coral sensitivity data, with *permissive* (*probable effects*) and *strict (possible effects)* thresholds based on ROC analysis (Fisher et al. 2018) and are the values that yield a 5% true positive and a 5% false positive error rate, respectively. The 80th and 95th percentile boxplots show the range of site level values for the corresponding percentiles calculated from the baseline time series data, excluding cyclones. Note that DLI is a negative stress metric (low values represent negative conditions), and is therefore based on the 20th and the 5th, rather than the 80th and 95th percentiles. The middle line of the boxes indicates the median value, upper and lower hinges the interquartile range, with the whiskers extending to the most extreme data point no more than 1.5 times this interquartile range.

The discrepancy between the observed coral tolerance limits of Fisher et al. (2018) and the ANZECC/ARMCANZ P_{50}–P_{80} approach raises the vexing and unresolved question (Fox 2016) of what should the percentile be to achieve the desired outcome. To explore this question the equivalent percentile values required to reach the *strict* (possible effects) and *permissive* (probable effects) threshold values for Barrow Island (based on estimates of probability of non-zero mortality during the dredging project). The calculated percentile values required to reach these thresholds are all exceptionally high, with the thresholds usually exceeding P_{99} of the baseline time series (Table 2). Only sites AHC and SBS (Figure 1) had baseline percentiles less than P_{99} for the *strict* threshold, and only site SBS had a baseline percentile less than the P_{99} for the *permissive* threshold for running 14 day mean turbidity (Table 2). Percentiles for running 14 day mean light stress were all above P_{99} for the *permissive* (probable effects) threshold, with a mean across sites of >0.99, with the mean percentile values for the light based *strict* (possible effects) threshold at 0.98 (Table 2).

Table 2. Estimated percentile values required to reach the *permissive* (probable effects) and *strict* (possible effect) thresholds for each site for the Barrow Island dredging project sites (see Figure 1).

	Permissive (probable effects) guideline		**strict (possible effects) guideline**	
Site	Turbidity $\log_{10}$(NTU+1)	Light stress: $1-(DLI/30)^{1/3}$	Turbidity: $\log_{10}$(NTU+1)	Light stress: $1-(DLI/30)^{1/3}$
AHC	>0.99	>0.99	0.97	0.92
LNG1			>0.99	0.94
MOF1				0.97
SBS	0.98		0.92	0.95
ANT, BAT, DUG, LNG0, LNG2, LNG3, MOF3	>0.99		>0.99	>0.99
DIW, ELS, LNGA, LNGB, LNGC, MOFA, MOFB, MOFC, REFN, REFS, TR		nd		nd
mean	0.996	0.999	0.993	0.979

Note that data during cyclone periods has been removed, and only sites with more than 360 days of baseline data for light, and 40 days of baseline data for turbidity are included (i.e. no data)

In conclusion, threshold derivation using high-order percentiles of natural (baseline) conditions have been proposed (ANZECC/ARMCANZ 2001, McArthur et al. 2002). The percentile analysis above shows that the

application of a P_{50}–P_{80} approach (ANZECC/ARMCANZ 2001) and the 95th percentile MacArthur (2002) approach are likely to be far too conservative than even the *strict* (possible effect) thresholds of Fisher et al. (2018), which in turn is based on highly conservative estimates of the probability of non-zero mortality of corals at Barrow Island. This would result in excessive Type 2 errors and based on the 2 week running mean period, only extreme conditions are needed (i.e. a P_{50}–$P_{>99}$). Whilst extreme (relative to baseline) disturbances in water quality can occur during dredging (Jones et al. 2015a), and it is theoretically possible to use this approach, estimating percentiles at such extreme tails of the distribution (P_{98}, P_{99} and P_{100} percentiles) is not advisable, as small events captured (or not captured) during the baseline time series may substantially alter the resulting guidelines.

3.3 Multiplier based threshold extrapolation

Another more general approach that could be used to estimate guideline values when no data are available on the sensitivity of benthic organisms, is to use a multiplier technique (i.e. multiplying the baseline levels by a certain value). As with the P_{50}–P_{80} approach above, the question is what should this multiplier be? In much the same way as the preceding analysis, we also explored this question using the Barrow Island water quality and coral health dataset by first calculating the mean ($\bar{x}$) and median (P_{50}) of the 14 d running mean turbidity ($\log_{10}$[NTU+1], equation 1) and light stress ([1–[DLI/30]$^{1/3}$, equation 2) during the baseline period for each site and the corresponding multiplier needed to reach the 14 d running mean *permissive* (probable effects) and s*trict* (possible effects) thresholds of (Fisher et al. 2018). The results were very similar across sites, with the *permissive* (probable effects) thresholds averaging ~3× the $\bar{x}$ and P_{50} baseline 14 d running mean turbidity and 1.6× the $\bar{x}$ and P_{50} 14 d running mean light stress (Table 3). *Strict* (possible effects) thresholds were correspondingly lower and averaged 2.2× the $\bar{x}$ baseline turbidity and 1.4× the $\bar{x}$ baseline light stress (Table 3).

Table 3. Estimated multiplier values required to reach the *permissive* (probable effects) and *strict* (possible effects) thresholds for each site for the Barrow Island data set, based on the mean ($\bar{x}$) and median of the baseline time series. Note that data during cyclone periods has been removed (see Section 2.3), and only sites with more than 360 days of baseline data for light stress and 40 days of turbidity data are included.

	Turbidity Log_{10}(NTU+1)				**Light stress 1-(DLI/30)$^{1/3}$**			
	Permissive (probable effects)		***Strict (possible effects)***		***Permissive (probable effects)***		***Strict (possible effects)***	
Site	P_{50}	$\bar{x}$	P_{50}	$\bar{x}$	P_{50}	$\bar{x}$	P_{50}	$\bar{x}$
AHC	3.6	3.1	2.7	2.3	1.5	1.5	1.3	1.3
ANT	3.2	3.2	2.4	2.4	1.8	1.8	1.6	1.5
BAT	3.2	3.1	2.4	2.4	1.9	1.9	1.6	1.6
DIW	2.2	2.2	1.6	1.7				
DUG	3.3	3.3	2.5	2.5	1.9	1.8	1.6	1.5
ELS	2.8	2.9	2.1	2.2				
LNG0	3.3	3.3	2.5	2.5	1.4	1.4	1.2	1.2
LNG1	3.2	3.1	2.4	2.4	1.4	1.4	1.2	1.2
LNG2	3.7	3.7	2.8	2.8	1.5	1.5	1.3	1.3
LNG3	3.1	3.1	2.3	2.3	1.6	1.6	1.4	1.4
LNGA	2.8	2.8	2.1	2.1				
LNGB	1.9	2.1	1.5	1.6				
LNGC	3.4	3.4	2.6	2.6				
MOF1	3.2	3.2	2.4	2.4	1.5	1.5	1.3	1.3
MOF3	3.3	3.1	2.5	2.4	1.7	1.7	1.5	1.4
MOFA	2.4	2.3	1.8	1.8				
MOFB	2.7	2.6	2.0	2.0				
MOFC	2.0	2.2	1.5	1.7				
REFN	4.5	4.5	3.4	3.4				
REFS	2.9	2.2	2.2	1.7				
SBS	2.6	2.5	1.9	1.9	1.8	1.7	1.6	1.5
TR	3.0	3.0	2.2	2.3				
Mean	**3.0**	**3.0**	**2.3**	**2.2**	**1.6**	**1.6**	**1.4**	**1.4**

Text Box 1. Statistical treatment of data for multiplier calculations

The statistical treatment of data has a fundamental influence on the interpretation of differences between baseline and dredge periods, as well as reference and impact locations, and this becomes very apparent when considering multiplier factors. Using NTU on a log scale (as was done for our turbidity pressure metric, Fisher et al. (2018)) creates a greater emphasis on small values, and decreases the differences of larger values. This means that on a log scale variation among the lower baseline turbidity values will seem larger than variation during dredging across dredge impacted and non-impacted sites. During baseline log10 (NTU+1) ranged from 0.23 to 0.54 across sites (a 2.3 fold difference on this scale) transforming to 0.70 and 2.5 NTU (a 3.5 fold difference) using the raw NTU scale. In contrast, during the first 203 days of dredging log10 (NTU+1) ranged from 0.31 to 1.16 (a 3.7 fold difference) across sites, transforming to 1.04 and 13.5 NTU (a 13 fold difference) on the scale of raw NTU.

Similarly, as light stress is capped at 1 (because it is impossible to have less than 0 light), differences among naturally low light sites during baseline and dredging will appear smaller than for high light sites, for which observed changes may appear bigger in the context of multipliers. During baseline light stress varied from 0.12 to 0.58 (a 4.8 fold difference) which is equivalent to 20.4 and 2.2 DLI (a 9 fold difference). During the first 203 days of dredging however, light stress varied between 0.33 and 0.78 (a 2.4 fold difference) across sites which is equivalent to a range of 9 and 0.32 DLI (a 28 fold difference).

From these calculations it is clear that the scaling of the data has a significant effect on the multiplier, with much larger multipliers apparent when on the original scale of NTU and DLI than on log10 (NTU+1) and light stress [1-(DLI/30)$^{1/3}$]. This highlights the need for consistent statistical treatment of data. Also apparent from these calculations is the fact that the differences among sites even during baseline and for the scaled data (a 2.3 and 4.8 fold difference for log10 (NTU+1) and light stress respectively) are larger than the average difference between each baseline and the estimated *Strict* (Log10 (NTU+1) ~ 2.3; light stress ~ 1.4) or even *Permissive* (Log10 (NTU+1) ~ 3.0; light stress ~ 1.6) threshold value (Table 3).

3.4 Spatial variation in background conditions

Water quality data has been collected associated with capital dredging operations at multiple sites across four broad locations in the Pilbara (see Figure 1). These datasets include extensive baseline (pre-dredging) phases and provide an opportunity to explore spatial variation in the natural background sediment and light regimes in this region. The four capital dredging projects occurred in very different environmental settings and whilst the Barrow Island project was in an offshore, clear-water environment, this was not the case at other locations (see Capital dredging projects in NW WA and water quality above, Figure 1). The baseline water quality characteristic of these projects have already been described in Fisher et al. (2015), Jones et al. (2015a) and Abdul Wahab et al. (2017) and have shown pronounced between and within-location natural differences in turbidity – and yet coral reefs can clearly live across a range of conditions. This high variation in the natural turbidity and light regime under which corals can occur poses a considerable challenge in the context of threshold development, but it is unclear how thresholds should be adjusted according to background conditions.

A cluster analysis using the 80th percentile of running mean turbidity (as NTU) for the baseline (pre- dredging) phase (excluding cyclone effected data) across the four dredging projects identified four broad clusters (labelled 1–4, Figure 5A) including (1) a *high turbidity* group (MAN, BLR, SMSB, HAT,MDR, PLR, and PWR from Cape Lambert; REFS from Barrow Island and WINI from Burrup Peninsula); (2) a *moderate turbidity* group that includes two sites from Onslow (ROLLER and WRNWD), four sites from Barrow Island (MOFC, LNGB, SBS and DIW), two sites from Burrup Peninsula (KGBY, CHC4), and most of the remaining Cape Lambert sites; (3) a *low turbidity* group that includes most of the remaining sites from the Onslow and Barrow Island, along with the ANGI site from Burrup Peninsula and DLI from Cape Lambert; and (4) a final *extremely low turbidity* group including only BESS from Onslow (Figure 5A).

For the 80th percentile running mean DLI there were three dominant clusters: (1) a *dark* group that included a mix of Barrow Island and Onslow sites that either tended towards high turbidity or were relatively deep; (2) a group of sites with *medium* baseline light levels that were either of shallower depth or characterized by low turbidity; and (3) a *light* group including BESS and AIRLIE from Onslow, which had both low, or extremely low turbidity and were reasonably shallow (Figure 5B).

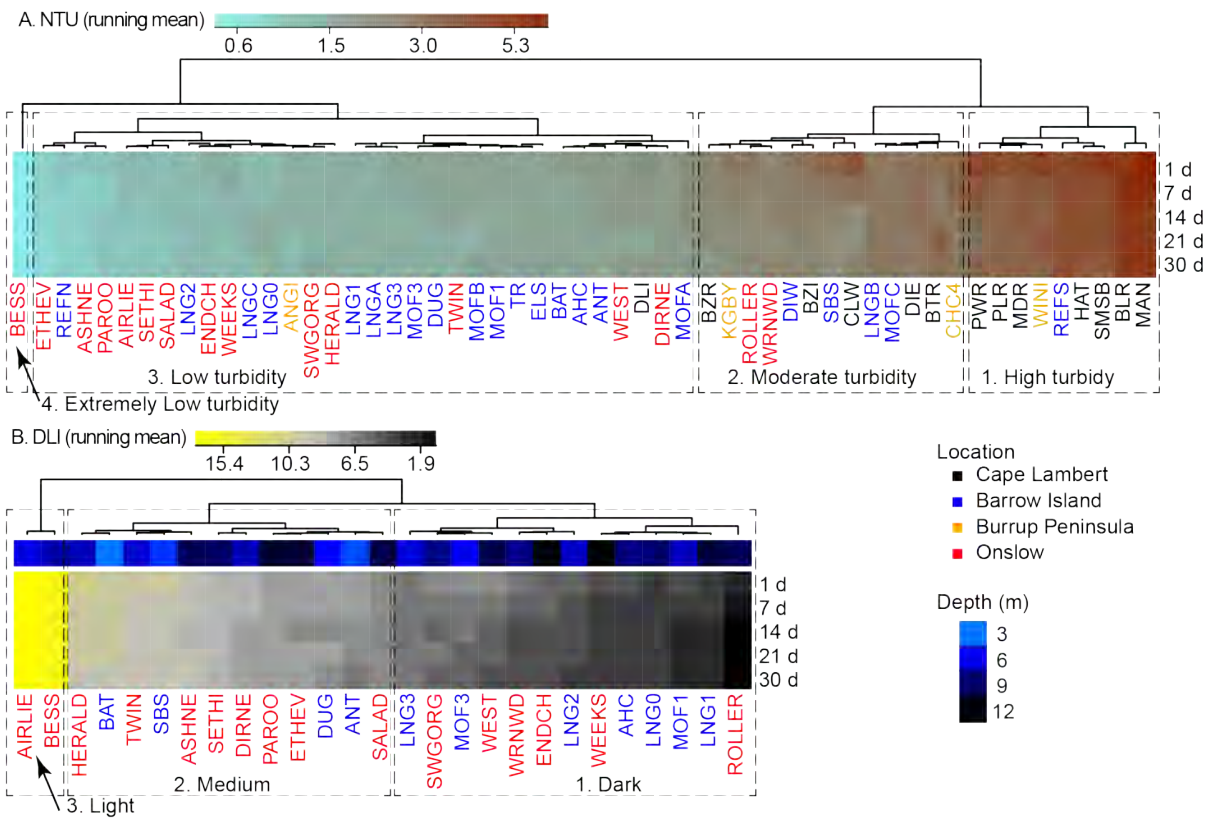


Figure 5. Cluster analysis and heat maps showing baseline NTU (A) and DLI (B) values, for a range of sites from four different dredging programs (see Figure 1). Dendrograms are based on the 80th percentile of the 1, 7, 14, 21 and 30 d day running means. Dashed boxes and associated numbers indicate the most significant groupings for NTU and DLI. Note that groupings are based on baseline time series only, excluding cyclone periods, and only sites having a minimum of 40 and 360 days for NTU and DLI respectively were included.

Using the $\bar{x}$ (across sites) multipliers for running 14 day mean turbidity and light stress from the Barrow Island project (Table 4), it is theoretically possible to estimate thresholds that may be appropriate for managing dredging across locations from different background (pre-dredging) turbidity regimes (Text Box 2). However, while the approach may work for sites from intermediate background conditions, it seems unlikely such extrapolated thresholds will be directly applicable for locations that deviate substantially from those typical at Barrow Island. For example, very extreme low NTU (~1.8 running 14 d mean NTU, *strict* (possible effect)) and high DLI (~3.7 running 14 d mean DLI, *strict* (possible effect)) thresholds are obtained for the extremely low turbidity and high light locations respectively (Text Box 2). It seems unlikely that mortality, or even sub-lethal stress, of corals will occur at such low turbidity and high light levels, even for very clear water locations. This is supported by several laboratory studies (Bessell-Browne et al. 2017b, Bessell-Browne et al. 2017d, Duckworth et al. In Prep) which failed to show evidence of impact at similar levels of exposure for corals collected from a relatively clear water environment on the Great Barrier Reef.

On the other end of the scale, extremely high thresholds are obtained for highly turbid locations (~38 running 14 d mean NTU, *permissive* (probable mortality)), and extremely low light thresholds occur for dark locations (~0.45 running 14 d mean DLI, *permissive* (probable mortality); Text Box 2). While it is possible that corals growing in such marginal environment are tolerant of such poor water quality conditions, experimental studies specifically exploring their capacity to cope would be required to definitively establish that such thresholds are appropriate in reality.

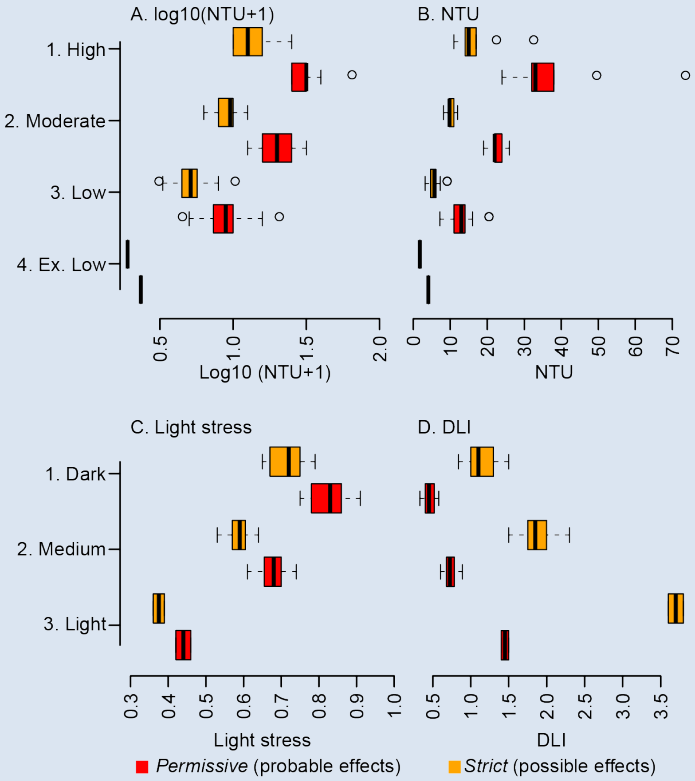
In addition to the problem of extremes, the analysis raises other issues. Firstly, the broad range of estimated thresholds across individual sites within each of the background groupings (see Text Box 2) highlights the difficulty in formally establishing appropriate groupings for applying specific management thresholds. The delineation between low, moderate and high turbidity sites, as well as dark, medium and light sites, is relatively

arbitrary, as in reality sites occur along a continuum of relatively poor to relatively good water quality conditions. Developing consistent guidelines or rules of thumb for assigning sites into particular natural sediment and/or light regimes is not trivial. Indeed, in the exploratory analysis leading to our final Figure 5 a wide range of possible groupings were generated, and these changed markedly depending on which percentiles and/or running means were used to summarise the water quality data included in the clustering. The issue is further complicated by the relative influence of extreme events, such as cyclones, on sediment dynamics at a given locations (see Figure 3). The groupings in Figure 5 were derived using data excluding cyclone periods and suggest substantial overlap between many of the Barrow Island and Onslow sites. However, Onslow was substantially more influenced by cyclone periods than Barrow Island (see Figure 3) and inclusion of cyclone data would suggest a greater differentiation between these two locations, with many of the inshore Onslow sites influenced by runoff from the Ashburton and/or high SSCs during these extreme weather events.

Text Box 2. Multiplier based estimated thresholds across locations with naturally varying turbidity and light regimes.

Using the mean (across sites) multipliers for running 14 day mean turbidity and light stress from the Barrow Island project (Table 4) we derived theoretical thresholds for managing dredging across locations of four different background (pre-dredging) turbidity regimes (groups 1-4, Figure 5A), and three different background light regimes (groups 1-3, Figure 5B).

In general, there was considerable variation among estimated thresholds across individual sites, even within each of the background groupings. This highlights the difficulty in formally establishing appropriate groups for applying specific management thresholds, as the delineation between low, moderate and high turbidity sites, as well as dark, medium and light sites, is relatively arbitrary, as sites generally sit along a continuum.



Depending on the background group the multiplier thresholds can vary widely. For the *extremely low* turbidity group (group 4, Figure 5A) running 14 d mean *permissive* (probable effects) thresholds for BESS from Onslow were 0.37 (or 4.1 NTU); and *strict* (possible effects) thresholds were only 0.28 (or 1.8 NTU). At the other extreme, for the *high* turbidity group (group 1, Figure 5A) running 14 d mean *permissive* (probable effects) thresholds were on average 1.5 (or 38 NTU) with a mean *strict* (possible effects) threshold of 1.1 (or 17 NTU).

For the *high* light group (group 3, Figure 5B) the running 14 d mean *permissive* (probable effect) threshold was 0.45 (or 1.45 DLI), with a mean *strict* (possible effect) threshold of 0.38 (or 3.7 DLI). For the *dark* group thresholds were correspondingly very low, with a mean *permissive* (probable effect) threshold of only 0.82 (or 0.45 DLI) and a mean *strict* (possible effect) threshold of only 0.72 (or 1.2 DLI).

Whether or not such extreme low and high thresholds apply in reality cannot be established with the available data, however it seems unlikely that these estimated thresholds would be appropriate to use for managing dredging impacts.

4 The relationship between turbidity, sediment deposition and coral health

The preceding sections have discussed elevated SSCs and reduced light availability as key pressure parameters but smothering of corals by high sediment deposition rates is recognized as one of the key cause effect pathways leading to mortality of adult corals (Jones et al. 2016) and has been shown to be particularly important for non-branching corals (Bessell-Browne 2017). Smothering results in decreased solutes (such as oxygen) and metabolite exchange, decreased filtering/feeding, and also loss of light. This leads to bleaching of the underlying tissues, and potentially focal and multifocal lesion formation and tissue mortality unless the sediments are washed away by a storm (Figure 6, Jones et al. (2016)). It frequently occurs during dredging programs and typically in close proximity to the excavation or dredge material placement (dumping) sites – see Figure 6 and images in (Bak 1978, Foster et al. 2010, Jones et al. 2015b, Jones et al. 2016, Bessell-Browne 2017, Duckworth et al. 2017).

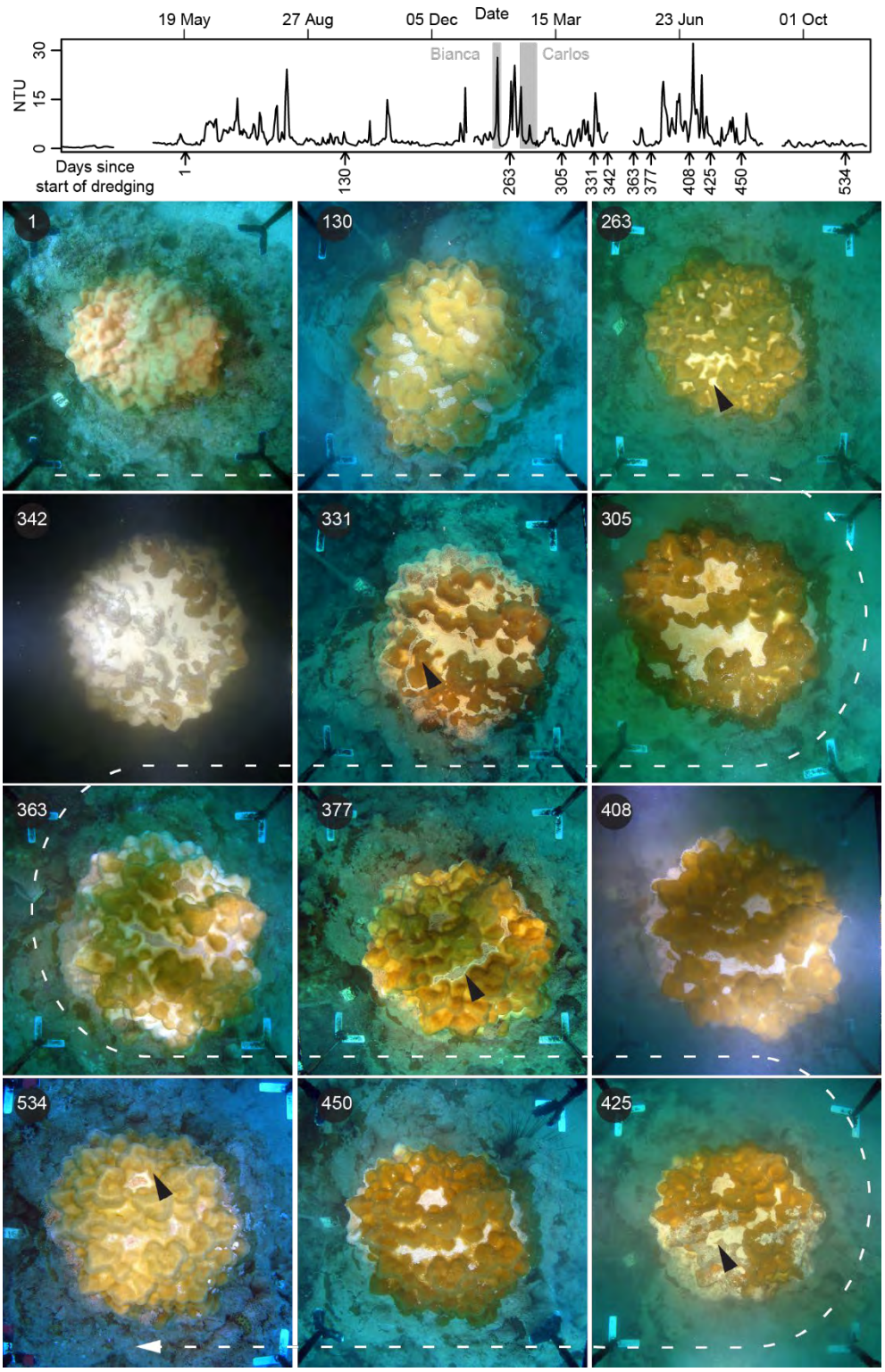


Figure 6. Time sequence of a single massive *Porites* colony from day 1 (20 May 2010) to day 534 (4 Nov 2011) at site LNG2 (see Figure 1) located 1 km away from the dredging during the 530 d Barrow Island project. The images show a build-up of sediment on low points (depressions) on the surface (day 130). Shortly after Tropical Cyclone Bianca (23–30 January 2011) accumulated sediment has been re-suspended from the surface leaving discoloured (bleached) areas of unhealthy but nevertheless live tissue (see arrow, day 263). More sediment accumulates in the now bleached depressions (days 305 & 331) and the colony responds by producing a mucous sheet (see arrow, day 331). After sediment accumulates again (day 342) and re-suspends, more extensive areas of bleached tissue in the depressions are evident (day 363). These bleached areas regained their pigmentation relatively quickly (day 377) and after producing another mucous sheet (see arrow, day 425) the colony has begun to re-grow tissue over the central lesion formed by the sediment deposition (see arrow, day 534).

4.1 Measuring sediment deposition

Estimating the sediment deposition zone near dredges and when deposition rates cause biological effects is critically important for predicting if and where coral mortality could occur (i.e. defining the spatial extent of any impacts). This is important information at both the EIA stage of a dredging project (where predictions are made of possible environmental effects) and also in monitoring programs associated with dredging phase, where proponents need to make sure they are compliant with the EIA predictions (EPA 2016).

The problem is that until recently (see Whinney et al. (2017a)), there have been no reliable techniques for measuring sedimentation rates at scales which are relevant to the physiology of local organisms (i.e. mg cm^{-2} d^{-1}) (Jones et al. 2016). Ridd et al. (2001) attempted to address this problem by developing an optical backscatter sediment accumulation sensor (SAS), which is effectively an upwards facing nephelometer capable of detecting sediment deposition in the laboratory in a quantitative way, and which also sometimes worked in the field (Thomas & Ridd 2005). The technique was used in the water quality monitoring program during the Barrow Island dredging project (Figure 1). Inspection of the *in situ* field-collected data showed that sediments settling on the flat glassy plate (where deposition measurements were made) frequently resuspended or possibly saltated over the surface (Stark et al. 2017). This resulted in low or sometimes spurious readings and the lack of a steady accumulation of sediment over the measuring period prevented calculation of a sediment surface density and sediment deposition rate as outlined in Thomas and Ridd (2005) and Whinney et al. (2017a). Despite the issues with the SAS instruments they nevertheless provided at least a semi-quantitative (i.e. relative rather than absolute) indication that sediment deposition was occurring at multiple sites at different distances from the dredging. Fisher et al. (2018) used this semi-quantitative data and found that deposition based exposure metrics were actually quite strong predictors of coral mortality during the Barrow Island dredging campaign further emphasizing the need to quantify sediment deposition as a key dredging-related pressure parameter.

These observations prompted a complete redesign and re-configuration of the sensor head, as recently described in Whinney et al. (2017a). It has yet to be deployed during dredging programs and until there is a better understanding of the likely ranges of sediment deposition associated with dredging projects, and the response of the underlying coral communities, only indirect mechanisms can be used to estimate sediment deposition as a pressure parameter.

High turbidity can be generated during natural resuspension events from tidal currents, swell and wind-driven waves (Larcombe et al. 1995, Ogston et al. 2004), but elevated turbidity this does not necessarily equate to high sediment deposition, as under these energetic conditions the water column hydrodynamics are sufficient to keep at least some of the sediments from settling. Sediment resettlement (deposition) will inevitably occur but during quiescent periods and following dilution, and over a period where energy in the water column is gradually decreasing. This effect was shown following the redesign of the SAS instrument heads and deployment in inshore turbidity reef systems of the GBR (Whinney et al. 2017a). The sensors were deployed just before a high wind, natural resuspension event where nephelometrically-derived SSCs exceeded very high levels (100 mg L^{-1}). The sediment deposition increased nearly 10 fold over the ~10 d duration of the turbidity event, and over the entire duration of the deployment largely mirrored the water column turbidity. However, on a finer (sub-daily) scale sediment deposition and turbidity were uncoupled. The deposition occurred either relatively uniformly (constant) or as pulsed short-term (4–6 h) periods of 'enhanced' deposition. When deposition occurred was dependent on tidal cycles and phases, as well as decreased wave orbital velocities associated with loss of the daily sea-breeze (Whinney et al. 2017a). Thus, a high turbidity level may result in different sediment deposition scenarios depending on whether the peak in turbidity coincides with a lower or higher energy water column.

In contrast to natural resuspension events, dredging activities can generate extremely high SSCs in a low energy water column where the hydrodynamics are insufficient to keep the sediments in suspension. This could lead to high sediment deposition rates and although deposition at any particular moment would be related to water column hydrodynamics, site and time specific effects could be averaged out when considering longer time periods. The turbidity (in units of NTU) was compared to the estimates of sediment accumulation from the sediment accumulation sensors (SAS Figure 7). Based on daily means the relationship between turbidity and

deposition (SAS data) or sediment coverage was very weak and wedge shaped (Figure 7A), with turbidity adequately predicting real high deposition days but falsely predicting many high deposition days that were not supported by the SAS data. The relationship between turbidity and the SAS data improves markedly when averaging is carried out at longer time frames, with a reasonably good relationship at 100 d timescales (Figure 7B, R^2=0.34), and a strong relationship when averaged across the entire time series at a site level (Figure 7C, R^2=0.68) between turbidity and the SAS data (Figure 7). Better relationships when averaging over longer time periods may also explain why longer duration thresholds for sediment deposition work better for predicting impacts to corals *in situ* than shorter time frames (Fisher et al. 2018).

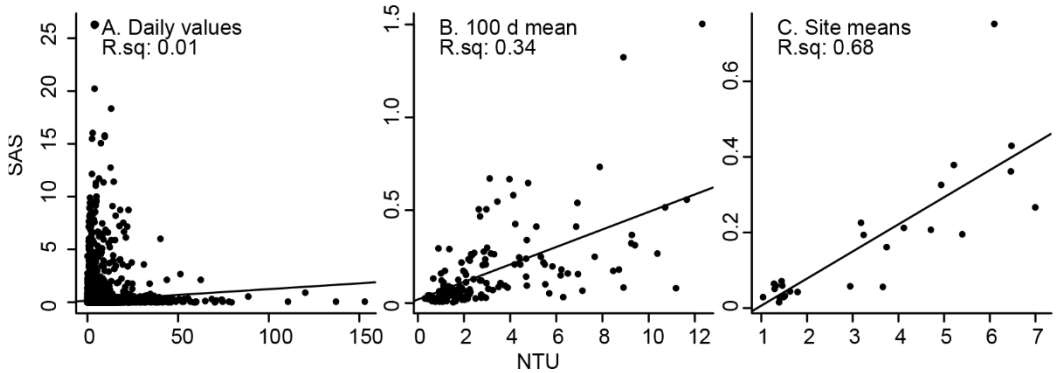
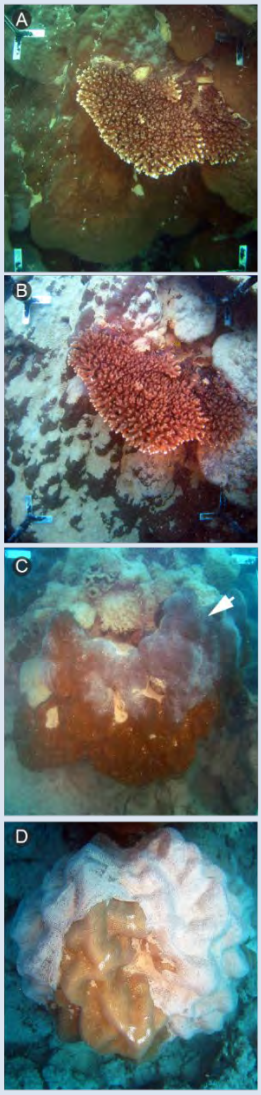


Figure 7. A-C. Pairwise correlations between sediment accumulation sensor (SAS) readings of settled surface sediment density (SSSD), sediment overburden estimates. Correlations are shown for daily means and as site level summaries across the whole time series.

In conclusion, sediment deposition resulting in smothering of corals is a key dredging related pressure, especially for non-branching coral morphologies. It is very difficult to measure *in situ,* although a newly developed sediment deposition sensor (see (Whinney et al. 2017a)) should prove useful in determining natural and dredging related sediment deposition levels and understanding coral tolerance levels. Until this has been established this interim analysis has shown that turbidity can be correlated with sediment deposition and semi-qualitative estimates of deposition (using the SAS techniques) – but only when averaged over longer rather that shorter time periods.

4.2 Relative distance of sediment related impacts

During the Barrow Island project there was a near continuous unidirectional southerly flow of currents (Evans et al. 2012) and water sampling was conducted along part of the gradient involving measurements of underwater light, turbidity, and semi-quantitative indices of sediment deposition with the SAS instrument (Fisher et al. 2018). Concurrent with these water quality parameters a variety of coral colony health measurements were made, including two bio indicators of sedimentation effects (sediment accumulation on the surfaces, Bessell-Browne (2017)); and mucus sheet production (Bessell-Browne et al. (2017a); see Text Box 3 and Figure 6), and the probability of non-zero mortality of massive *Porites* spp. corals (Fisher et al. 2018). This provides an opportunity to relate water quality characteristics (including sediment deposition) to coral health along a clear dredging related water quality gradient, and calculate distances of dredging related impact (Text Box 4).



Text Box 3. Bio-indicators of the effects of sediments on corals.

For corals sediment accumulation on their surfaces is undesirable, resulting in reduction in metabolite exchange, filtering/feeding and light for their algal symbionts. This can lead to bleaching of the underlying tissues, and potentially partial mortality (focal and multifocal lesion formation) unless the sediments are washed away for example by a storm (Figure 6, Jones et al. 2016).

Some corals, particularly branching species with their more sieve like morphologies, are highly adept at self-cleaning and shifting sediments. Other morphologies such as encrusting and massive *Porites* spp. – and especially those with local depressions (minima) on their surfaces – are much less capable of shifting sediments which then become trapped and begin to accumulate over time. The contrasting sediment shifting abilities of the branching versus massive growth form is shown opposite in images A and B taken *in situ* during the Barrow Island project at a site ~1 km away from the dredging activities – showing smothering of the underlying *Porites* spp. but no sediment accumulation on the tabulate *Acropora* sp.

Hemispherical or dome shaped (massive) growth forms *Porites* spp. such as *Porites lutea* and *P. lobata* appear to have another strategy to cope with occasional periods of high sediment loads. They occasionally produce mucous sheets (see C) that can envelop the entire colony and which trap settling particulate material such as sediment, eventually sloughing off the surface (see D), leaving relatively clean and sediment-free tissue underneath. As discussed in (Bessell-Browne et al. 2017a), there are multiple lines of evidence from field observations and controlled laboratory-based studies suggesting an association between mucous sheet production and sediments. The ability to occasionally clear their surfaces of settled material could be one reason why these *Porites* spp. are capable of thriving across a range of environments including nearshore, more turbid coastal waters. Sheet formation and sloughing allows colonies to occasionally clear their surfaces of settled material and, as a protective reflex, it is functionally analogous to a human sternutation (sneeze).

Both the presence of surface accumulation of sediments and mucus sheet prevalence are useful indicators of sediment exposure.

Text Box 4. Fitting distance decay relationships to the water quality and coral health metrics at Barrow Island.

Distance decay relationships were fitted for a range of water quality and coral health metrics across the strong dredging related pressure gradient that occurred during the dredging project at Barrow Island. Relationships were fit using $\log_{10}$ distance from dredging activity as the predictor. Distance was included on a log scale because previous work had already demonstrated an exponential decay relationship between distance from dredging and changes to water quality conditions (Fisher et al. 2015) and exploratory analysis indicated this non-linearity was evident across all the water quality and coral health variables examined.

Water quality summary metrics were based on the worst case (P_{100}) for each site observed in the first 203 days of the dredging program (i.e. before the bleaching event), to remain consistent with (Fisher et al. 2018). As mucus covering on corals was quantified on a categorical scale (see Bessel-Browne et al. 2016) these categories were converted to their equivalent proportional midpoints and averaged to the site level for analyses. The mean proportion of colonies with sediment covering in excess of any mucus covering was also calculated, and indicates where sedimentation has exceeded coral's sediment clearing capacity. To ensure all decay relationships were comparable to Fisher et al (2018) only the first 203 days of observations were used in the site level means.

In order to calculate 10% 'Effect Distances' (ED_{10} – akin to the EC_{10} and LD_{10} commonly used in ecotoxicology), all response variables were rescaled to between 0 and 1 before analysis, with 0 representing the minimum and 1 representing the maximum observed values. All relationships were fit as logistic regressions using Bayesian generalised mixed models using

JAGS (Plummer 2003) and the R2JAGS (Su & Yajima 2015) package in R (R Core Team 2016) with uninformative priors. Estimated 95% credible bands were calculated from five MCMC chains (60000 burn-in, 90,000 iterations and a thinning of 5). Site was included in all models as an additional observation-level random effect to allow for over-dispersion of the binomial distribution (Bolker et al. 2009, Harrison 2014), as preliminary analyses indicated this was necessary to obtain adequate fits.

There were strong relationships with distance from the dredging activity for all of the metrics examined, although the rate of change with distance varied. For NTU 14 d running mean the estimated distance of 10% effect (ED_{10}, see Text Box 4) occurred at 28 Km (Figure 8). The analysis supports previous distance analysis of this data which showed measurable perturbations in water quality in a southerly direction at Barrow Island as far as 19.6 Km (Fisher et al. 2015). In contrast, dredging-related elevations in sediment deposition dissipated rapidly, with an estimated ED_{10} of only 7.7 Km (based on a worst case 60 d running mean of the SAS instrument readings, Figure 8). Sediment related coral health exposure metrics dissipated even more rapidly than even the SAS measurements of disposition, with ED_{10} values for the percentage of colonies with accumulated sediment and mucus on massive *Porites* spp at only 3.1 km (Figure 8). As might be expected, elevation in the probability of non-zero mortality of massive *Porites* spp we even closer to the source of dredging than either of these sub-lethal metrics, with an upper estimate of a 50% probability at only 1.4 km (Figure 8, ED_{50}).

Distances of exceedance of the 14 d running mean turbidity and 60 d running mean SAS measurement thresholds correspond well with the observed biological effects, with *permissive* thresholds representing probable effects very close to the dredging activity (0.42 and 0.67 km respectively) and the *strict* thresholds representing possible effects reaching distances of up to ~4 km), Figure 8). This highlights that while elevations in turbidity may occur at considerable distances, the distances of thresholds likely to cause coral mortality are very close to dredging.

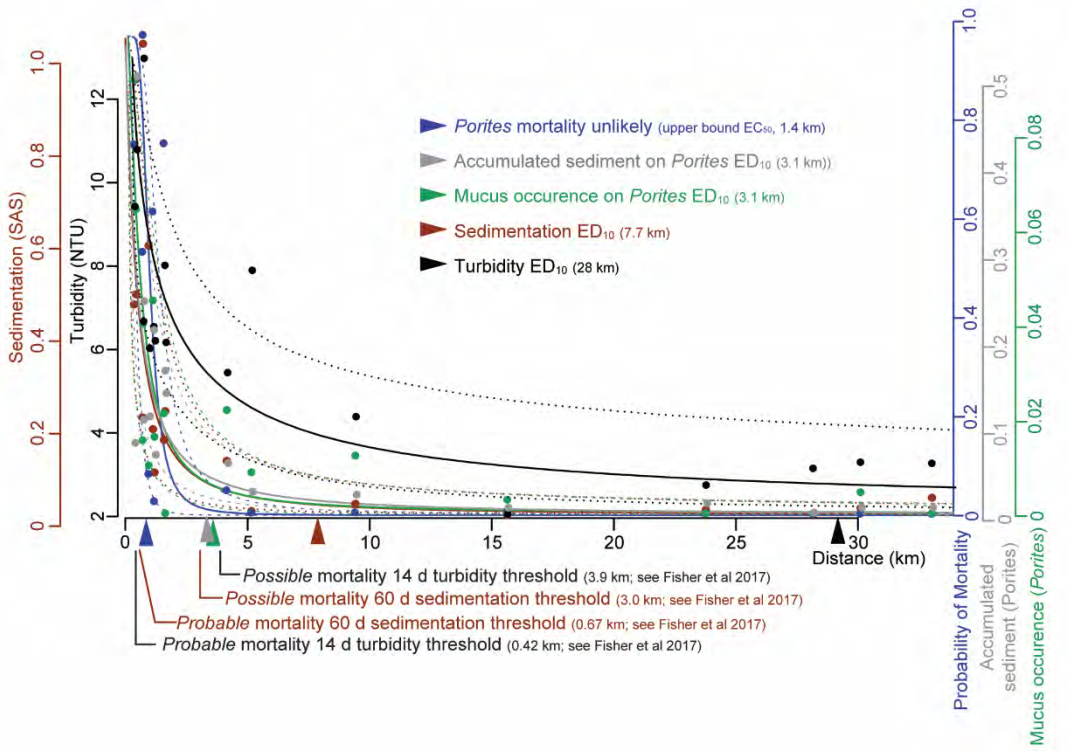


Figure 8. Distance decay relationships for turbidity and sediment accumulation (SAS technology) for the first 203 days of dredging during the Barrow Island project, as well as proportional mucus sheet coverage, proportion of colonies with accumulated sediment (in excess of mucus covered sediment) and probability of mortality of *Porites* spp. All response variables examined were rescaled to between 0 and 1 before analysis and fitted using logistic regression with $\log_{10}$ distance from dredging activity as the predictor. See Text Box 4 for statistical details. Coloured arrows at the base of the plot indicate estimated 'Effect Distances' of 10% (ED_{10}). Lines below the x-axis indicate distances of exceedance of threshold values, including both the *permissive* (probable effects) and *strict* (effects possible) thresholds from (Fisher et al. 2018).

The water quality and coral health measurements conducted along the north to south water quality gradient (Figure 1) did sample slightly to the side of the primary axis of flow of the suspended sediments from the dredging areas (as there were no coral reefs in the area immediately in the direction of the dredge plume) based on the analysis of satellite images by Evans et al. (2012). This means that the distances of actual sediment deposition and extremely high turbidity may extend further than what can be estimated on the observed data, thus the absolute distances are not as important as the relative distances of the different pressure parameters. Nevertheless, the analyses show that while turbid plumes may travel considerable distances and affect turbidity at distances of ~20 km (see also Fisher et al. (2015)), coral mortality and effects associated with sediment deposition (smothering) occur considerably closer to the dredging activity.

The good agreement between the many field-based observations of sediment accumulation on *Porites* spp and mucus sheet prevalence, the SAS measurements, and mortality in *Porites* spp. provides strong evidence that for morphologies with flat planes of tissue where sediments can accumulate in local surface depressions and cannot be moved by active and passive processes, sediment deposition is one of the key pressure parameters associated with dredging activities.

These field based observations of elevated deposited sediments on *Porites* spp colonies are supported by the laboratory based analyses conducted in Theme 4.6, where the sediment rejection ability of 8 coral species and 3 morphologies were assessed in several experiments with different sediment types, particle sizes, flow regimes and sediment deposition rates. The study showed that the branching species, *P. damicornis* and *A. millepora* were highly capable of self-cleaning and removing sediments compared to the foliose and massive morphologies. The species tested were capable under a slight flow of removing all sediment up to 20 mg cm^{-2} d^{-1} leaving only small residual deposits typically less than a few percentage of the surface area. The massive (dome shaped corals)

morphologies could do the same up to 40 mg cm^{-2} d^{-1} and the branching species *A. millepora* managed to clear a sedimentation rate (under static conditions) of up to 235 mg cm^{-2} d^{-1}. Although these numbers cannot be related back to conditions *in situ* for the reasons described previously, they show how capable the branching morphologies are at removing sediments, providing explanations for the images in Text Box 3. The laboratory-based studies also showed that once corals were smothered in sediments bleaching of the tissues under the sediment occurred in as little as 4 days, but no mortality was observed even after 6 weeks following this smothering (Duckworth et al. 2017). These laboratory observations of sediment-induced bleaching are similar to the field based observations (see Figure 6 day 263 and day 363).

5 The relationship between turbidity, light attenuation and coral health

In contrast to the massive *Porites* spp. branching species such as the Acroporids and Pocilloporids, with their more sieve-like architectures are highly effective at shifting sediments, and live tissues were rarely smothered in sediment ({Jones, In prep #1017}, see also Figure 9 and Text Box 3). It is likely that other pressure parameters, such as light reduction, and/or in combination with elevated SSCs (Bessell-Browne et al. 2017b), are more important than sediment deposition for these morphologies.

The water quality conditions measured during the Barrow Island project (Jones et al. 2015, Fisher et al. 2016) were used to guide a series of laboratory-based studies of cause-effect pathways and dose-response relationships, also providing quantitative information on the response of corals of different species and morphologies to the elevated SSCs and light reduction tested both alone and in combination (see Theme 4.6, Bessell-Browne et al. 2017a, Bessell-Browne et al. 2017). Initial studies showed that there were no negative effects on corals (*Acropora millepora*, *Pocillopora* spp. *Montipora aequituberculata*, and *Porites* spp) after 28 d exposure to very high SSCs (up to 100 mg L^{-1}), providing that light levels were sufficient (>8 mol photons m^{-2} d^{-1}). This suggested it is the light attenuating properties of suspended sediments that is the main pressure parameter (for suspended sediment) – although some interactive effects with SSCs were noted at the very high concentrations.

These results prompted a similar experiment examining the low light tolerance of *A. millepora*, *Pocillopora* spp and to crustose coralline algae (CCA) over 28 d (in the absence of elevated SSCs). The results showed exposure to very low light (<0.1 mol photons m^{-2} d^{-1}) caused bleaching in the corals (the dissociation of the symbiosis) and bleaching and partial mortality of the CCA, yielding 30 d EC_{10} thresholds (irradiance which results in a 10% change in colour) of 1–2 mol photons m^{-2} d^{-1}. Subsequent longer term experiments were performed with combinations of elevated SSCs and associated light reduction showing *A. millepora*, *Turbinaria mesenterina* and *Porites* spp could survive and grow over a ~40 d period in a combination of 2.2 mol quanta m^2 d^{-1} and 10 mg L^{-1}, whilst some sub-lethal effects were noted in *P. damicornis* at this combination. At higher combinations (30 mg L^{-1} and a DLI of 0.3 mol photons m^{-2} d^{-1} and 100 mg L^{-1} and a DLI of 0 mol $photon^{-2}$ d^{-1}) bleaching, loss of lipids, and partial mortality become common for all species. Overall, the most sensitive species appeared to be the branching species *P. damicornis* > *A. millepora* followed by the massive corals *Porites* spp. The foliose coral *T. reniformis* was the most tolerant species, which is consistent with its reputation as a turbid water specialist.

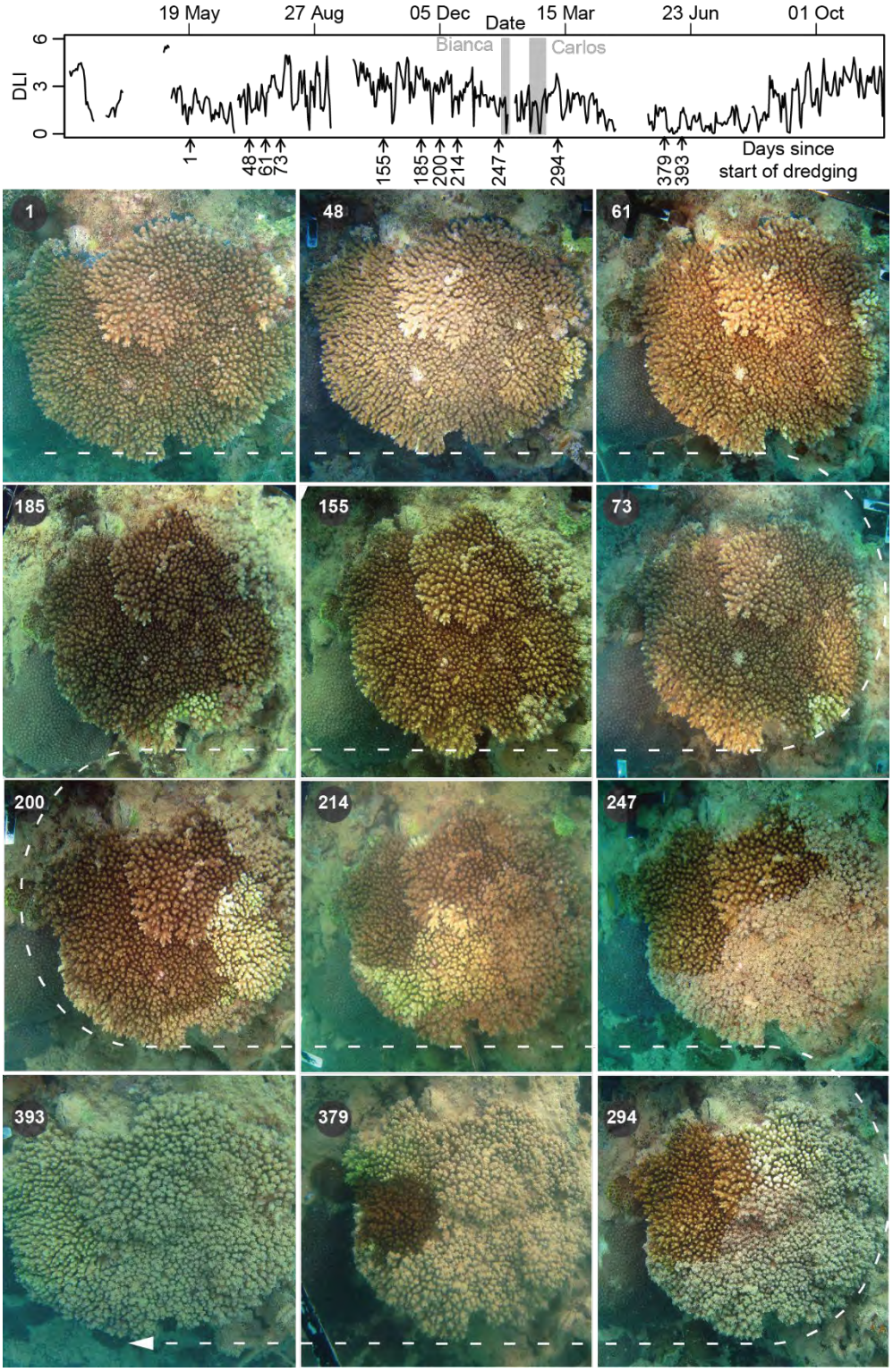


Figure 9. Time sequence of a single tabulate *Acropora* spp. from day 1 (20 May 2010) to day 393 (16 June 2011) at site MOF1 (see Figure 1) located 0.8 km away from the dredging during the 530 d Barrow Island project. The images show the gradual increase in partial mortality of the living (brown-coloured) tissues leaving a white, tissue-free skeleton what becomes fouled with a filamentous green algae. The dead areas become covered with sediments (grey colouration) over the ~300 d it took for the colony to suffer whole-colony mortality.

Similarly to sedimentation impacts, we explored the distance decay relationships of light-based water quality metrics relative to the probability of observing non-zero branching *Acropora* and Pocilloporidae mortality. Like turbidity, impacts on the light climate can occur at distances very far from dredging, with estimated 10% effect distances (ED_{10}) occurring at 31 km for 14 day running mean DLI (Figure 10). However, impacts on branching *Acropora* spp. only occurred very near the dredging activities, with an upper estimated distance of 50% probability of non-zero mortality (equivalent to the NEC) at only 1.9 km (Figure 10). That light conditions detrimental to coral health only occur very close to dredging are further supported by the distances of exceedance of both the field- and laboratory- based thresholds (Figure 10). Over the 203 days examined 30 day running mean DLI never fell below the laboratory derived 50% effect concentration (EC_{50}) threshold for coral mortality, and a 10% threshold (EC_{10}) was only exceeded at distances less than 1 km). This corresponds very well with estimated distances of exceedance of field based *permissive* 14 d running mean DLI and NTU thresholds indicative of probable coral mortality effects (Fisher et al. 2018), which were only 0.21 and 0.42 km respectively).

Laboratory based thresholds for sub-lethal impacts to coral, such as an estimated EC_{10} for coral bleaching were exceeded at greater distances than lethal impacts (3.4 km), as were laboratory based estimates for crustose coralline algae (EC_{50} CCA, 3.1 km, Figure 10). Non-lethal impacts to coral up to 3-4 km from dredging is consistent with estimated distances of exceedance of field based *strict* 14 d running mean DLI and NTU thresholds, indicative of possible coral mortality effects (Fisher et al. 2018), which were 2.7 and 3.9 km respectively).

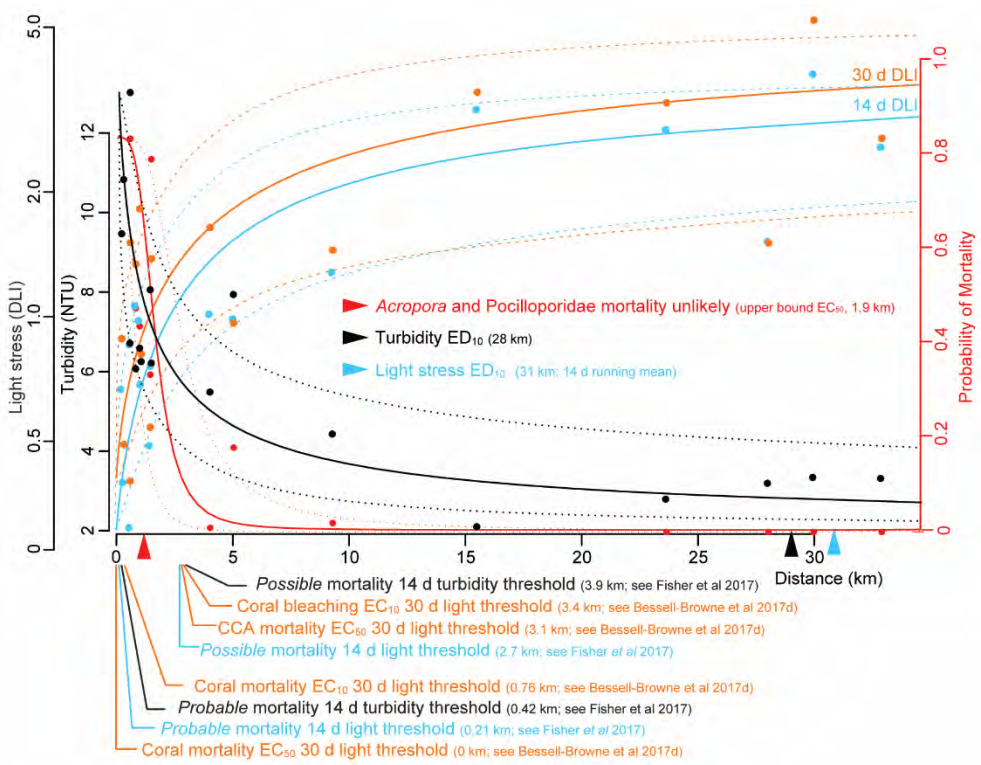


Figure 10. Distance decay relationships for benthic light availability derived for the first 203 days of dredging during the Barrow Island project, as well as mortality in branching *Acropora* spp. All response variables examined were rescaled to between 0 and 1 before analysis and fitted as logistic regressions with log_{10} distance from dredging activity as the predictor. See Text Box 4 for statistical details. Coloured arrows at the base of the plot indicate estimated 'Effect Distances' of 10% (ED_{10}). Lines indicate distances of exceedance of threshold values, including *permissive* (probable effects) and *strict* (effects possible) thresholds from (Fisher et al. 2018), as well as laboratory based EC_{50} and EC_{10} 30 d thresholds for the probability of coral mortality (Bessell-Browne et al. 2017d), laboratory based EC_{50} for crustose coralline algae (CCA) mortality, and a laboratory based EC_{10} value for elevated coral bleaching (Bessell-Browne et al. 2017d).

6 Synthesis and Discussion

Characterising water quality hazards for impact prediction purposes (at the EIA stage) and management of operations during dredging is a complex issue. There are a myriad of ways water quality measurements can be summarised into metrics of dredging related pressure. The preceding analyses have shown that for shallow water corals there are different cause-effect pathways for different morphologies acting either alone or in combination with other pressures.

The key or dominant pressure parameters are likely to vary differently between coral morphologies. In corals that have flatter surfaces and with surface depressions where sediments can become trapped and accumulate, smothering is probably the dominant cause-effect pathway. *Porites* spp. which are susceptible to smothering can produce mucous sheets, and the sloughing of sediment-laden sheets can temporarily clean the surfaces of accumulated sediments. Nevertheless, if deposition rates are too high, or for other species without alternative surface cleaning capabilities, the sediment smothering is likely to result in loss of light to the underlying tissues and lead to bleaching and ultimately lesion formation (partial mortality) in several days or weeks. If the accumulated sediments are removed by water motion the lesions can eventually heal, but until the dead areas are grown over with live tissue, the areas will continue to trap and accumulate sediments. In branching species, which are much more capable of freely shifting depositing sediments even at high sediment loads, the dominant effects are most likely associated with the light reduction. The results of laboratory based studies examining the effects of light deprivation supported the spatial analysis at Barrow Island, indicating that most of these effects are occurring in areas of high turbidity close to the dredging activities in the areas where sediment smothering is a problem for some morphologies.

Although these conclusions do not rule out interactive effects of the different pressure parameters (i.e. light limitation affecting deposition prone morphologies), or the continual need for self-cleaning in branching species in response to contact with high SSCs, they suggest that that zones of high impact based on different cause-effect pathways (sediment deposition and light attenuation) are similar, which simplifies impact prediction and modelling for these diverse coral reef communities.

6.1 Delineating Zones of Impact

Combining the different sources of evidence for the distances of likely lethal, and sub-lethal, impacts to coral through both sedimentation (see Figure 8) and light attenuation (see Figure 10) pressures, it is possible to delineate zones of impact for Barrow Island (Figure 11). We define a zone of potential high impact (ZoHI, (EPA 2016), 0–1.9 km) that encompasses distances of exceedance of experimentally derived light based thresholds for increases in the probability of mortality of corals (EC_{10}, Bessell-Browne et al. (2017d)), previously calculated thresholds corresponding to a high probability of observing non-zero coral mortality (see *Permissive* thresholds NTU and DLI (Fisher et al. 2018), and estimated distances where massive *Porites* and branching *Acropora* spp mortality are now unlikely (Figure 11). This ZoHI is followed by a Zone of Moderate Impact (ZoMI, 2.9–4 km) that encompasses the region immediately outside the ZoHI up to distances where sub-lethal impact thresholds (30 day light based EC_{10} threshold for coral bleaching), or thresholds indicative of only *possible* coral mortality (*strict* thresholds, Fisher et al 2018) are no longer exceeded (Figure 11). Beyond the ZoMI there is an additional Zone of Influence, where changes to water quality may occur but there are no dangers to underlying coral communities (Figure 11).

The estimated distances at which mucus cover and proportion of colonies with accumulated sediment on *Porites* colonies becomes negligible (EC_{10}) occur within the ZoMI, and may prove to be a useful indicator within this zone. The presence of mucus on *Porites* spp. is considered a sub-lethal response as the *Porites* spp. can slough the mucus layer and entrapped sediments, clearing their surfaces. However, long term sediment accumulating on *Porites* spp. colonies is more serious as it will lead to multifocal lesion formation (pathological discontinuities of tissue) and tissue mortality of the corals (see above).

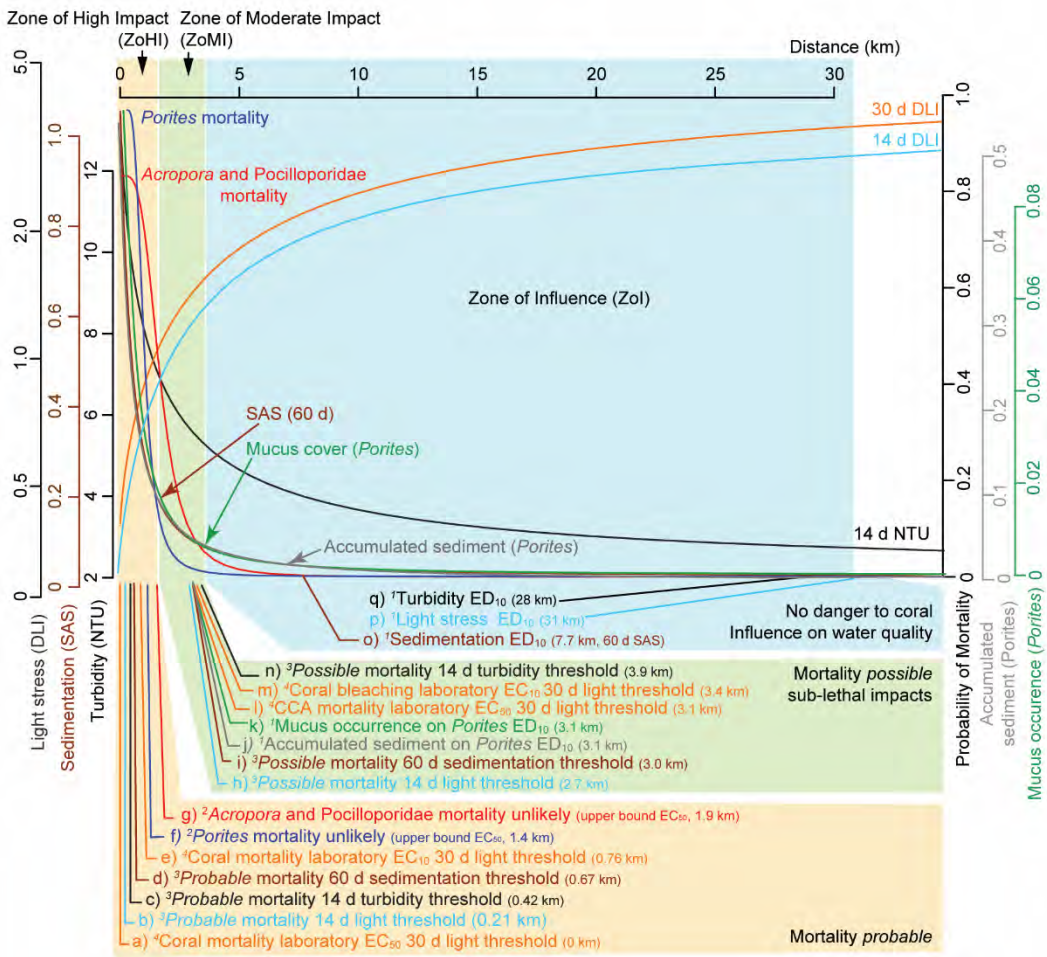


Figure 11. Aggregate distance decay plot for a range of sedimentation and light related water quality and coral health metrics for the first 203 days of dredging during the Barrow Island project. All response variables examined were rescaled to between 0 and 1 and fitted using logistic regression with $\log_{10}$ distance from dredging activity as the predictor (See Text Box 4 for statistical details). Three zones of impact are defined based on the multiple lines of evidence provided, including: a Zone of High Impact (ZoHI) that encompasses all estimated distances likely to result in elevated coral mortality; a Zone of Moderate Impact (ZoMI) that encompasses all estimated distances likely to lead to sub-lethal impacts on coral communities and *possible* non-zero coral mortality; and a Zone of Influence (ZoI) in which only elevations to water quality metrics occur with no danger to coral. Coloured lines at the base of the plot indicate estimated distances of effect from one of four sources: 1) estimated distances of 10 % effect (ED_{10}, from the maximum to the minimum predicted value at the furthest distance based on the fitted logistic curve, akin to an EC_{10} or LD_{10} in ecotoxicology); 2) estimate upper bound distances where *Porites* massive or *Acropora*/Pocilloporidae branching mortality is unlikely (<50% probability - NEC, ED_{50}); 3) distances of exceedance of field based threshold values (from Fisher et al. 2018); and 4) distances of exceedance of laboratory based threshold values (from Bessell-Browne 2017d). Individual letters are as follows: a) Coral mortality EC_{50} 30 d light threshold (Bessell-Browne et al 2017d); b) *Probable* mortality 14 d light threshold (Permissive - Fisher et al 2018); c) Probable mortality 14 d turbidity threshold (Permissive - Fisher et al 2018); d) Probable mortality 60 d sedimentation threshold (Permissive - Fisher et al 2018); e) Coral mortality EC_{10} 30 d light threshold (Bessell-Browne et al 2017d); f) *Porites* mortality unlikely (upper bound ED_{50}); g) *Acropora* / Pocilloporidae mortality unlikely (upper bound ED_{50}); h) *Possible* mortality 14 d light threshold (Strict - Fisher et al 2018); i) *Possible* mortality 60 d sedimentation threshold (Strict - Fisher et al 2018); j) Accumulated sediment on *Porites* ED_{10}; k) Mucus occurrence on *Porites* ED_{10}; l) Crustose coralline algae (CCA) mortality EC_{50} 30 d light threshold (Bessell-Browne et al 2017d); m) Coral bleaching EC_{10} 30 d light threshold (Bessell-Browne et al 2017d); n) Possible mortality 14 d turbidity threshold (Strict - Fisher et al 2018); o) Sedimentation ED_{10}; p) Light stress ED_{10}; and q) Turbidity ED_{10}.

Delineating the inside boundary of the ZoMI using the *permissive* thresholds and upper bound estimates for where mortality is unlikely is potentially conservative, as these indicate only high probabilities of *non-zero* coral loss. The approach could be easily modified to establish thresholds based on probabilities of mortality effects greater than zero to reflect the boundary of the ZoMI (e.g. a 30% relative decline). However, actual mortality estimates were relatively low during the first 203 days of dredging activity at Barrow Island, with no sites having >30% relative mortality. Thus for shorter term dredging programs (<203 days), and in the absence of extremely long term or other acute cumulative stressors, the worst case reported conditions during the first 203 days of the Gorgon project at Barrow Island may be considered as appropriate 'concentrations of <30% effect'. Using the conservative approach of non-zero mortality however, ensures that any lethal impacts to corals are well contained within ZoHI and that the resulting spatial zoning is environmentally robust.

Changes in the ecological functioning of reef systems may be altered by dredging activities at greater distances than those observed for coral mortality, because other taxa can be relatively more sensitive and because there can also be sub-lethal impacts to coral health that may not necessarily result in direct mortality of corals. At Barrow Island sub-lethal impacts on corals, such as low level coral bleaching (30d light based EC_{10} threshold for coral bleaching), changes to the surrounding Crustose Coralline Algae (CCA) communities, or very low probabilities of coral mortality (see Strict thresholds for SSSD, NTU and DLI) are not expected further than distances of ~4 km (Figure 11). These distances also correspond to the distance at which the bulk of sedimentation related effects have dissipated, and can be considered the distance at which we move from the zone of influence on water quality in towards a zone of moderate impact on coral communities (Figure 11). While direct coral mortality is unlikely within this zone, sub-lethal impacts and changes in the underlying coral community may have important implications for these coral communities if the impacts remain over the long term. For example, like corals, CCA are an essential structural component of coral reef ecosystems (Littler et al. 1985, Steneck 1986, Maudsley 1990) and provide chemical cues for settlement of many benthic invertebrate larvae, including corals (Johnson & Sutton 1994, Morse et al. 1996, Heyward & Negri 1999). Reduced prevalence of CCA associated with low light conditions during dredging may result in declines in coral recruitment; potentially exacerbating issues associated with increased deposited sediments interfering with coral settlement (Ricardo et al. 2017).

In summary, there are multiple lines of evidence to suggest that thresholds representing probable (*permissive)* and possible (*strict*) mortality of corals can be used to delineate the ZoHI/ZoMI boundary and ZoMI/ZoI boundaries respectively.

6.2 Triggering management action

During dredging operations triggers are often used to initiate appropriate management actions to ensure that the desired environmental protection outcomes are achieved. Management actions can include those aimed at improving water quality conditions, such as reducing production rates, relocating the dredge, initiating dredge overflow management, or evening stopping dredging altogether if environmental impacts may result in non-compliance with Ministerial conditions. Triggers for different management actions will depend implicitly on the spatial zone in which the trigger is exceeded (see 6.1 above), as they should relate in a direct way to the thresholds that are used for delineating zones of impact. Thus the distinction between thresholds for probable (p*ermissive)* and possible (*strict*) mortality of corals above will directly apply as management triggers within the appropriate spatial zone (Table 4). The management actions that the *possible* and *probable* mortality thresholds should initiate vary among each zone (Table 4).

Table 4. Relationship between thresholds for possible and probable mortality and indicative management actions that may be taken given an exceedance within different management zones.

	Zone of High Impact (ZoHI)	**Zone of Moderate Impact (ZoMI)**	**Zone of Influence (ZoI)**	**Outside Zone of Influence**

Possible mortality (*strict* threshold)	No action	Monitor operations	Ensure improved water quality	Stop dredging
Probable mortality (*permissive* threshold)	Monitor operations	Ensure improved water quality	Stop dredging	Stop dredging

Within the ZoHI total coral loss is generally allowed and there is no reason to act following exceedance of *possible* mortality thresholds. Exceedance of *probable* mortality thresholds may trigger active monitoring of operations. Within the ZoMI exceedance of *possible* mortality should only result in the monitoring of operations, whereas exceedance of *probable* mortality thresholds should initiate actions to ensure improved water quality conditions. Within the ZoMI exceedance of *probable* mortality thresholds would not require halting dredging, assuming that some mortality in the ZoMI is allowable under the project conditions. Within the ZoI exceedance of *possible* mortality thresholds should initiate management actions guaranteed to improve water quality conditions, and exceeding *probable* mortality thresholds should result in immediate halting of dredging activities and initiation of benthic monitoring to assess if there has been non-compliance within the ZoI.

6.3 Thresholds for impact prediction and management

While the concept of *probable* and *possible* mortality thresholds link to both the impact predictions used to define spatial zones during the EIA assessment stage, as well as the management triggers to be used within each zone during dredging operations, the question remains as to what are the best metrics and thresholds to use? Should these be based on suspended sediment concentrations (SSCs and their surrogate, NTUs), light reduction or sedimentation? To add further complication various lines of evidence suggest that both acute and chronic stress play a role in compromising coral health during dredging (Bessell-Browne 2017, Fisher et al. 2018), and that corals of different morphologies are sensitive to both light and sedimentation related effects (Bessell-Browne 2017, Bessell-Browne et al. 2017b, Duckworth et al. 2017), with the direct effects of suspended sediments *per se* being much less (Bessell-Browne et al. 2017b).

A range of thresholds have been derived using the coral mortality data from Barrow Island (see Fisher et al. (2018)), as well as various laboratory based thresholds (Bessell-Browne et al. 2017d). While these thresholds are useful, distilling the pressure field generated during dredging down to a few thresholds representing only one duration or baseline exceedance intensity is problematic for several reasons. Firstly, there are strong correlations between all water quality derived dredging related pressure metrics and the relative importance of different metrics cannot be dis-entangled (see Text Box 5). Furthermore, taking single threshold values from the overall pressure field may not properly account for potential acute (or conversely chronic effects). To illustrate, consider a dredge management protocol based on not exceeding a strict 14-day running mean turbidity threshold of 5.3 NTU. Conceivably, it may be possible to maintain water quality conditions below such a running mean whilst potentially exceeding the baseline 80th percentile of 1.8 NTU for the entire dredging phase. However, exceedance based thresholds suggest an exceedance of 45% of days during dredging may compromise coral health (Fisher et al. 2018).

Text Box 5. Correlations among dredging related pressure metrics in the field

		SAS			Turbidity			Light stress		
		60 d running mean	% exceed.; p80	consec. exceed.; p95	14 d running mean	% exceed.; p80	consec. exceed.; p95	14 d running mean	% exceed.; p80	consec. exceed.; p80
SAS	60 d running mean	1	0.9	0.9	0.8	0.8	0.7	0.6	0.6	0.4
	% exceed.; p80	0.9	1	0.8	0.8	0.9	0.7	0.7	0.8	0.7
	consec. exceed.; p95	0.9	0.8	1	0.7	0.7	0.5	0.6	0.7	0.6
Turbidity	14 d running mean	0.8	0.8	0.7	1	0.9	0.9	0.8	0.8	0.6
	% exceed.; p80	0.8	0.9	0.7	0.9	1	0.8	0.9	0.8	0.6
	consec. exceed.; p95	0.7	0.7	0.5	0.9	0.8	1	0.7	0.7	0.7
Light stress	14 d running mean	0.6	0.7	0.6	0.8	0.9	0.7	1	0.8	0.7
	% exceed.; p80	0.6	0.8	0.7	0.8	0.8	0.7	0.8	1	0.8
	condec. exceed.; p80	0.4	0.7	0.6	0.6	0.6	0.7	0.7	0.8	1

Exposure metric correlations for nine dredging related pressure metrics representing the best (see Fisher et al. (2018)) across each of three metric types (running mean; percentage exceedance; and consecutive exceedance) for each of three cause-effect pathways of mortality (sedimentation – SSSD, suspended sediments – Turbidity and reduced light – Light stress).

Strong correlations among pressure metrics in the field demonstrates the inter-relatedness of the pressure pathways, making it is impossible to be sure of the best pressure metric time-scale, cause-effect pathway, and the associated thresholds to recommend for impact assessment modelling and dredging management. This is because there is no way to dis-entangle the importance of chronic versus acute impacts, as these all occur concurrently during a dredging campaign. While one metric may perform statistically slightly better than another, this is in the context of the co-occurring conditions defined by the other water quality pressure metrics.

The most robust approach for managing the impacts of future dredging projects is to ensure that water quality conditions remain within safe bounds across the full hazard profile in the context of both cumulative probabilities of exposure as well as running mean values across different temporal scales, with various thresholds for impact inferred from dredging related hazard profiles known to be harmful to corals. While direct empirical data are available for hazard profiles associated with light reduction (shown in Figure 12), hazard profiles for sedimentation are problematic because of the issues associated with measuring ecologically relevant sedimentation rates in the field (Whinney et al. 2017a). Although sediment deposition is a key cause-effect pathway it cannot be measured *in situ*. In the interim SSC (and/or NTU, shown in Figure 12) are suggested as a means of defining hazard profiles that delineate safe and un-safe conditions (to coral) in the hope that this will represent a profile(s) that is also safe/un-safe in the context of deposition, as over the long term there is a good relationship between the two (see Section 4).

Dredging related hazard profiles for SSC/NTU and DLI show clear relationships with distance from dredging activities, and in Figure 12 the delineation between potentially harmful profiles (<1.9 km, upper distance for potentially harmful effects, orange area, Figure 11) and safe profiles (> 1.9 km, see Figure 11) can be seen as the colour difference between the very near 'red and black' sites and the more distant 'blue and green' sites (Figure 12). There is an area of overlap between some more distant 'blue' sites and some of the very near 'red' sites. It is this overlap which relates to the uncertainty associated with deriving thresholds where the probability density function of pressure metrics overlap between impact and non-impact locations (Fisher et al. 2018). A line delineating the very lowest range of 'red' potential impact sites for SSC/NTU indicates where impacts are *possible* but not guaranteed, as some safe 'blue' sites sit above this line (Figure 12). In contrast, a line delineating the very upper range of blue sites indicates conditions that now exceed all known benign sites, and suggest that impacts to coral are now *probable* (see annotations b and d, Figure 12).

By fitting non-linear quantile regressions to the lower bounds of all sites <1.9 km (unsafe, 'red-black' sites) and upper bounds of all sites >1.9 km (safe, 'blue-green' sites) from the dredging (see smooth dark black lines, Figure 12) it is possible to obtain corresponding *possibl*e effects (dashed black lines) and *probable* effects (solid black lines) thresholds across the full hazard profile for both the cumulative probability distribution, as well as across

all durations of running means (Figure 12, Table 5).

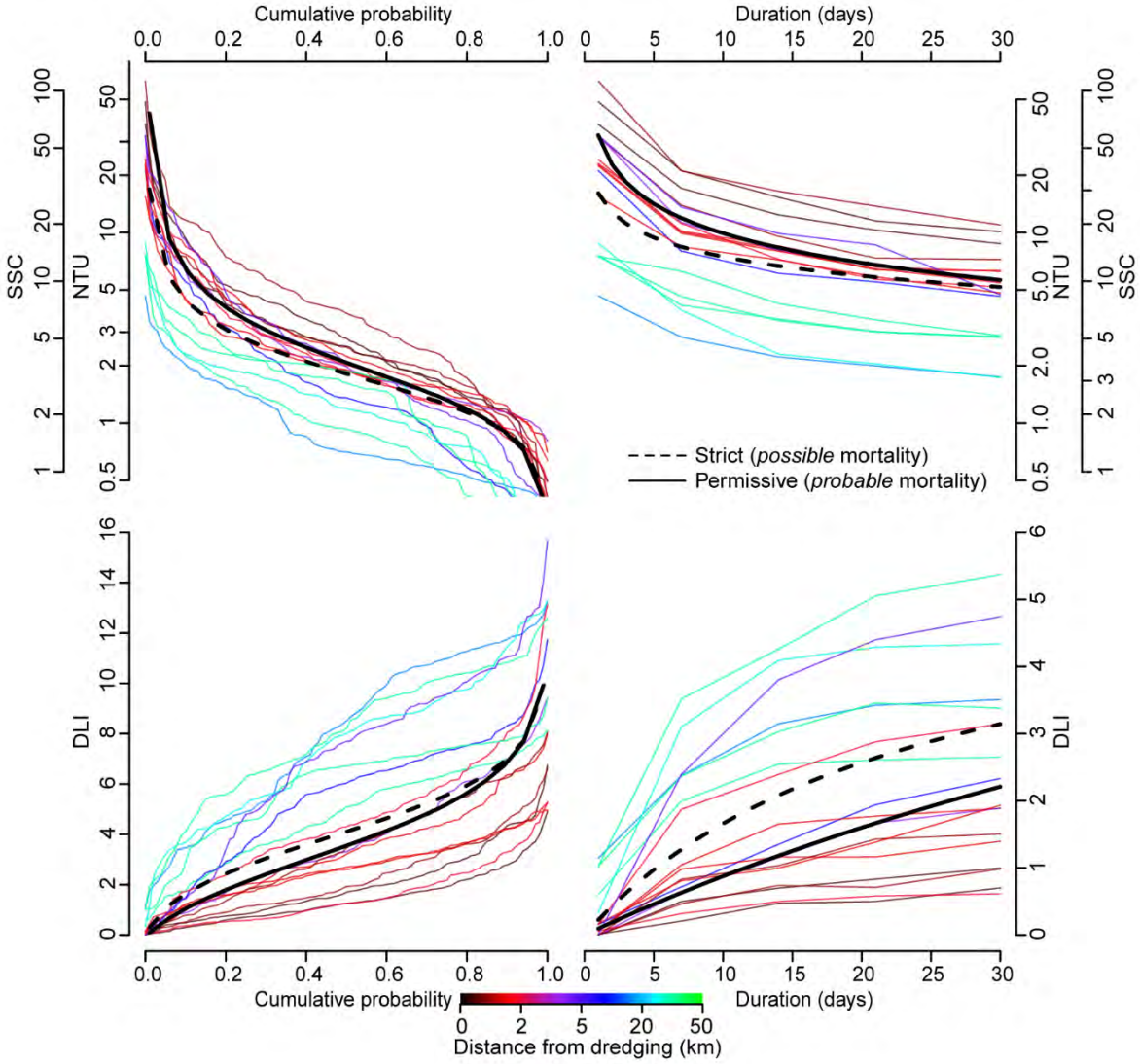


Figure 12. Exposure profiles for cumulative probability (left hand plots) and worst case running means from 1 to 30 days duration (right hand plots) for SSCs (mg L^{-1} based on an approximate conversion of SSC=1.8 × NTU, upper plots) and DLI (mol quanta m^{-2} d^{-1}, lower plots) for the first 203 days of dredging at Barrow Island. Coloured lines show individual profiles across 17 sites of varying distances to the dredging activity, with distance from dredging indicated by the colour bar. The interface between acceptable (blue and green lines, will not cause mortality of corals), versus harmful (black and red, may cause coral mortality) hazard profiles can be used to infer the conditions (across the entire hazard profiles) known to be harmful to corals. Cumulative probability, *possible* mortality fitted lines are given by the following equations: NTU = 10^0.450*exp(-0.222*log(CP/(1 − CP)))-1 and DLI = (exp(-1.84 * exp(-0.199 * log(CP/(1 − CP))))/(1+exp(-1.84 * exp(-0.199 * log(CP/(1 − CP))))))*30. Cumulative probability, *probable* mortality fitted lines are given by the following equations: NTU = 10^0.488*exp(-0.263*log(CP/(1 − CP)))-1 and DLI = (exp(-2.01 * exp(-0.228 * log(CP/(1 − CP))))/(1+exp(-2.01 * exp(-0.228 * log(CP/(1 − CP))))))*30. Running mean, *possible* mortality fitted lines are given by the following equations: NTU = 16.1* exp(-0.333 * log(Duration)) and DLI = (0.235*Duration)/(1+(0.0414*Duration)). Running mean, *probable* mortality fitted lines are given by the following equations: NTU = 32.4* exp(-0.514* log(Duration)) and DLI = (0.097*Duration)/(1+(0.0104*Duration)).

Table 5. Derived *possible* and *probable* coral mortality thresholds across the complete hazard profiles for cumulative probability (A) and running mean durations (B) for managing dredging activities near coral reefs. *Possible* effects (aka strict) thresholds are interpolated from the lower bounds of all sites <1.9 km (unsafe, 'red-black' sites, upper distance for potentially harmful effects, orange area Figure 11) and *probable* effects (aka permissive) thresholds are interpolated from the upper bounds of all sites >1.9 km (safe, 'blue-green' sites) from the dredging (see smooth dark black lines, Figure 12). Interpolations at the upper and lower bounds were achieved via non-linear 0.9 and 0.1 quantile regressions. SSC units are mg L^{-1} and DLI units are mol quanta m^2 d^{-1}.

Thresholds type		Possible effects *(strict)*			Probable effects *(permissive)*		
		>NTU	>SSC	<DLI	>NTU	>SSC	<DLI
Cumulative probability (% days)	**90%**	0.7	1.3	7.4	0.7	1.3	6.6
	80%	1.0	1.9	6.3	1.2	2.1	5.4
	70%	1.3	2.3	5.5	1.6	2.8	4.6
	60%	1.5	2.8	4.9	1.9	3.4	4.0
	50%	1.8	3.2	4.4	2.3	4.1	3.4
	40%	2.0	3.7	3.8	2.6	4.8	2.8
	30%	2.3	4.2	3.2	3.1	5.6	2.3
	20%	2.8	5.0	2.6	3.8	6.9	1.7
	10%	3.5	6.3	1.7	5.1	9.1	1.0
Running mean (days)	**1 d**	15.5	27.9	0.4	32.4	58.3	0.1
	3 d	**10.8**	**19.4**	**1.1**	**19.9**	**35.7**	**0.3**
	7 d	**8.2**	**14.7**	**1.8**	**13.6**	**24.5**	**0.6**
	10 d	7.3	13.1	2.2	11.6	20.9	0.9
	14 d	**6.5**	**11.7**	**2.5**	**10.0**	**18.0**	**1.1**
	17 d	6.1	11.0	2.7	9.2	16.5	1.3
	21 d	5.7	10.2	2.9	8.3	15.0	1.5
	28 d	**5.2**	**9.3**	**3.1**	**7.3**	**13.2**	**1.8**
	30 d	5.1	9.1	3.1	7.1	12.8	1.9

At the EIA stage, during impact prediction, both cumulative and running mean strict (dashed lines) and permissive (continuous lines) can be used to interrogate the coupled sediment transport and hydrodynamic models that predict sediment transport and fate, to estimate whether effects on corals are possible and/or probable respectively. If the predicted (modelled) hazard profiles cross the lines at any stage this would be used to categorize a site (ZoHI and ZoMI or ZoI). If the relationship between the sediment types and particle size and light attenuation properties are known (see WAMSI DSN Theme 3), then the DLI data can also be used in combination with the SSC data for impact prediction and site classification according to the zoning scheme.

During dredging programs 1–28 d running mean SSCs or DLI values can be plotted together with the guideline values, giving dredging proponents information on whether effects are possible or probable at both short term (days) and longer term (weeks) intervals. For example, we show nephelometrically derived SSCs (mg L^{-1}) collected at a water quality monitoring station ~200 m from dredging during the 530 d Barrow Island project (Figure 13), with 1–28 d running mean values over two × 10 d periods during the dredging where there were clearly defined short duration increases (spikes) in turbidity (Figure 13B and C). In the first period the mean daily SSC increased from 10 mg L^{-1} to 20–40 mg L^{-1} on days 3–6, close to or exceeding the 'possible thresholds' value, but over the remaining 4 days decreased to <5 mg L^{-1}. As a result, the 14 d running mean value was less than the possible effect guideline (11.7 mg L^{-1}, see Table 5, Figure 13B). In the second and much more substantial increase in turbidity, the mean daily SSC increased exceeded 50 mg L^{-1} on days 1–4 exceeding the 'possible' and 'probable' thresholds' values (Figure 13C). From day 5 onwards the SSCs decreased but were still >10 mg L^{-1} for the remainder of the period. As a result of the short term (acute) period of exceptionally high SSCs (>100 mg L^{-1}) and the sustained 10 d period of SSCs >10 mg L^{-1}, the 14 d running mean value exceed the possible (11.7 mg L^{-1}) and probable (18 mg L^{-1}) mortality thresholds considerably (see Table 5, Figure 13C). For contextual purposes the site was <200 m from the dredging and the analysis in Figure 3A shows it included the highest 1 day mean SSC in the 530 d dredging program. Similar peaks were observed at other sites and this spike is likely to have been caused by a storm combined with resuspension of loose unconsolidated sediments.

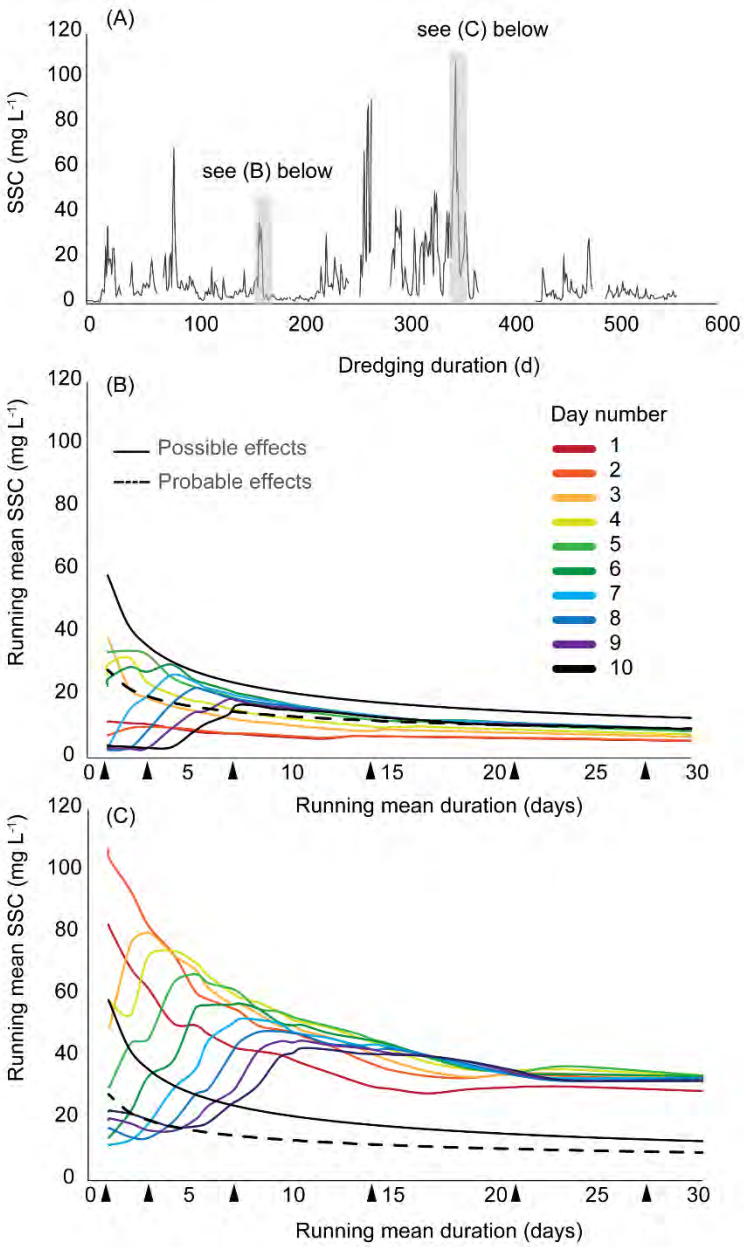


Figure 13. (A) Nephelometrically derived SSCs (mg L^{-1}) at a water quality monitoring site ~200 m from dredging during the Barrow Island dredging project. (B) and (C) show the 1–28 d running mean values calculated for each of 10 days spanning two turbidity events, one starting on day 166 (1 Nov 2010, B) and the other on day 349 (3 May 2011, C). The solid and dashed black lines show the derived *Possible* and *Probabl*e effect thresholds across the 1-28 d hazard profile (See Fig 12 and Table 5).

One of the primary considerations associated with development of the guidelines and the running means approach is that the total pressure must be accounted for, i.e.:

- Dredging-related pressure is calculated for light and SSCs in absolute terms (i.e. a running mean of $\boldsymbol{x}$ mg L^{-1} (or mol quanta m^2 d^{-1}) over a period $\mathbf{y}$ days), and not as days exceeding a certain value (i.e. number of days >$\boldsymbol{x}$ mg L^{-1} (or <x mol quanta m^2 d^{-1})) which would not allow determination of total load as how far the threshold was exceeded by is not known;
- Pressure is calculated as a running mean in absolute terms (i.e. $\boldsymbol{x}$ mg L^{-1}) and not relative to reference sites (i.e. $\boldsymbol{x}$ mg L^{-1} over a reference site value of $\mathbf{y}$ mg L^{-1}). From a physiological perspective corals do not discriminate between dredging-related and natural resuspension events and basing thresholds in comparison to reference sites would also not allow the total accumulated pressure to be monitored over the time frame considered.

- Calculating dredging-related pressure in absolute terms over clearly specified running mean time intervals allows more precise calculation of the accumulated load than coarser time averaging approaches – for example >x mg L^{-1} >20% of the time (where multiple consecutive exceedances may have much greater consequences than multiple exceedances interspersed with periods of otherwise good water quality).

Impacts of dredging on water quality are highly ephemeral and spatially complex, with changes to water quality occurring across both acute (changes to the extremes) and chronic scales (changes to long term conditions, Fisher et al. (2015), Jones et al. (2015a)). Time is an important consideration in threshold development and running means analysis (coupled to the possible and probable guidelines) have temporal components explicitly integrated into the analysis at multiple (rather than single) time periods accounting for both acute and more chronic effects.

Maintenance dredging projects in Western Australia are typically 4–10 weeks, slightly longer than in Queensland where the average maintenance dredging duration is generally 2–4 weeks (Ports Australia 2014a). The duration of capital dredging projects in Australia has ranged from a few weeks to up to 2.5 years, although most extend for 4–6 months; however, even within extended capital dredging projects, dredging often occurs across multiple locations (for example along a channel) and dredging pressures may occur on individual areas for a much shorter time. Thus basing thresholds on a 1–28 d time frame is likely to be functionally useful for both shorter (maintenance) and longer (capital) dredging campaigns.

We suggest that across the 1–28 d running mean period additional significance should be given to 3, 7, 14 and 28 days and special emphasis placed on the 14 time period, for the following reasons:

- Fisher et al. (2018), calculated running means across 1–60 day time scales for the Barrow Island data set and found that a 14-day running mean showed the strongest relationships with the probability of non-zero coral mortality for both turbidity and light stress;
- The time-frames allow dredging proponents enough time to plan and enact adaptive management activities if required (such as moving dredges, changing dredging modes or production rates etc);
- Having a comparatively short time component associated with a threshold (i.e. a running mean over weeks as opposed to months) means that if all dredging activities were reduced or to cease entirely, the running mean values would respond (i.e. improve) relatively quickly (as can be seen in Figure 13B);
- Natural events such as storms can be reliably predicted out to around 7 d by the Australian Bureau of Meteorology Next Generation Forecast and Warning System, and natural turbidity generating events such as storms or weather fronts are thus more foreseeable;
- Similarly, if there are cyclical periods associated with turbidity and light availability (such as with the 14 d spring-neap cycles), proponents can factor natural hydrodynamics into forward planning and decision-making purposes.

From an operational perspective, depending on the zone, and also the risk aversion of the proponent, crossing the possible or probable profiles (at any point), could result in management-considerations (such as changing dredging modes or production rates etc, moving dredges, to stopping all dredging). For example, exceeding a DLI or SSC *possible* effect running-mean threshold over any time period in sites designated as within a ZoI should probably prompt a management action. We suggest a management aim would be to be compliant to the zone at the 7, 14 and 28 day time frames even though the thresholds may be crossed intermittently between these times.

6.4 Caveats to threshold application and final considerations

The cumulative probabilities and running means analyses used here to characterize and manage hazard profiles during dredging are both intuitive and computationally simple, and the approach can be used with other organisms, although these will require different guidelines values (seagrasses, filter feeders, fish etc); however, there are some caveats associated with the thresholds and guidelines developed here. It has been emphasized that one of the key cause-effect pathways is sediment deposition, and this has yet to be accurately characterized in dredging conditions, or even during natural turbidity events. Elevated turbidity was used as a proxy for

sediment deposition on the basis that dredging re-suspends sediments into low energy water columns where sediment should fall rapidly out of suspension compared to natural resuspension events. Despite the recent development of instrumentation that can measured sediment deposition at appropriate scale (see Whinney et al. (2017b)), until there is better understanding of the relationship between deposition and coral smothering then one of the key cause-effect pathways is being measured by proxy.

The thresholds presented here are empirically tied to the Barrow Island project, a clear water high diversity shallow water coral reef ecosystem, and the laboratory-based experiments in turn were based on the pressures characterized during the Barrow Island project and conducted on specimens collected and maintained under similar conditions. Although the approach of turbidity and light hazard profiling and using a running means analyses during the dredging phase is suitable to equivalent environments, the absolute levels may not be applicable to more marginal reef sites such as turbid reef zones and deeper reefs. In WA there are different communities of corals in more turbid areas that generally contain a larger proportion of sediment and/or low light tolerant taxa, and it is difficult to know how the thresholds should or can be modified for such communities. The dynamics of light limitation means that typical depth zonation patterns on reefs are compressed into a much shallower areas, and light attenuation is more likely to be the dominant cause-effect pathway in naturally turbid reefs. Further uncertainty arises because, while it may be tempting to adjust thresholds to high levels for these theoretically more resilient communities, such communities are potentially marginal and may be naturally living much closer to their physiological limits. Further studies exploring variation in the sensitivity of individual taxa and communities to dredging related pressures would provide insight into the extent to which the derived thresholds may apply more generally, or if location specific thresholds are required.

Corals are a highly diverse group that differs substantially in terms of their life history traits, including polyp size, reproductive characteristics and morphology. They are known to also vary in terms of their susceptibility to thermal stress, and our experimental and *in situ* data also suggest they vary substantially in response to dredging related stressors. Models attempting to understand the temporal and spatial scales of potential impacts of dredging on nearby coral communities require clear data on the relative sensitivity of the constituent species. To make modelling of communities tractable, it is essential that functional trait groups are delineated. These groups must capture not only the key biological traits important in the context of population modelling, but also sediment and light stress sensitivities. Such traits could include the surface rugosity of colonies, polyp size, ability to switch between autotrophy and heterotrophy, along with the amount of energy reserves. Functional trait groups that capture coral sensitivities to dredging related pressures will be useful in helping guide management recommendations for different coral communities, as well as simplify modelling highly speciose, complex coral communities.

The knowledge gained through the “predictive links” monitoring undertaken in addition to the required compliance monitoring at Barrow Island is substantial. Science is an iterative process and similar efforts, which represent only small additional costs once extensive compliance monitoring is underway, should be encouraged wherever practicable for future dredging campaigns. Each new dredging project provides an opportunity to test the generality of thresholds and explore further complexities in the appropriate management of dredging operations in the vicinity of coral reef habitat.

7 References

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