Comparisons of benthic filter feeder communities before and after a large-scale capital dredging program

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WAMSI Dredging Science Node

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

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

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

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

Image 2: Photograph of mixed filter feeder community at the Onslow study site. (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: On deck photograph of the massive-cryptic sponge, Ciocalypta tyleri, freshly collected from the Onslow study site, which was one of ~600 biological specimens deposited to the Western Australian Museum (Source: AIMS)
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Executive summary

Filter feeder communities can be highly diverse and are important marine habitats in northwestern Australia where they often dominate the benthos at some locations. Sponges (Porifera) form significant components of these macrobenthic filter feeder grounds. The published literature clearly shows that sponges are influenced by sediment in a variety of ways. Most studies confer that sponges are able to tolerate, and in some cases thrive, in environments subject to sedimentation. However, relatively little is known on how they respond to dredging-related stresses, where there may be short term (acute) periods of poor water quality, and/or longer term (chronic) periods, superimposed on natural events. This study was undertaken to improve our understanding of the responses of filter feeder communities (focusing on sponges) to dredging related pressures, including elevated suspended sediment concentrations, light attenuation and sediment deposition. A secondary and important component of the work was a taxonomic study of sponge species biodiversity to improve knowledge for the Integrated Marine and Coastal Regionalisation for Australia (IMCRA) Pilbara Nearshore bioregion. A detailed catalogue of in situ and surface photographs of sponges was developed as a practical resource for future marine environmental studies in the area.

Capital dredging associated with the Wheatstone project occurred near Onslow (Pilbara region of Western Australia), and was WA’s largest marine dredging campaign to date. Multiple types of dredges were used, working often simultaneously and near continuously in multiple locations over an extended 2 year period. Dredging involved the excavation of a ~16 km entrance channel to a coastal LNG processing facility and sediment excavation to lay a gas trunk line. In the nearshore environment, the dredging was undertaken to create berth pockets, a material offloading facility and turning basins. In total ~31.4 Mm$^3$ of sediment was relocated to offshore dredge material placement sites (spoil grounds) and ~70% of the dredging was associated with the shipping channel.

Surveys of filter feeder communities were conducted in nearshore and offshore environments in March 2013, a few weeks before dredging started (pre-dredging survey), and in July 2015 a few months after dredging finished (post-dredging survey). Surveys included large-scale transects using video cameras towed behind a research vessel, which provided broad scale assessments of the benthic habitats. Surveys also included finer-scale studies using SCUBA diving, which also facilitated the collection of sponges for detailed taxonomic investigations and assessments of relative species abundance and chlorophyll content. A novel scoring system was used based on sponge functional growth form, which could provide more information on the susceptibility of sponge morphology to turbidity and sediment deposition. Water quality data collected by the proponent at 16 locations before and during the dredging were analysed to examine temporal patterns and the effects of dredging on turbidity (as nephelometric turbidity unit [NTU]) and light availability. The particle size distribution (PSD) of sediments was also collected by the proponent at 56 stations before, and 1-month and 7-months after dredging, and was analysed to examine deposition fields associated with the dredging. The surveys were concentrated on the entrance channel and in 3 areas — an Inner (nearshore) zone, a Middle (transition) zone and an Outer (offshore) zone, as determined by the baseline water quality and PSD analyses.

The study area is naturally turbid, with sites located in the nearshore environment (<5 km from the coast) up to 6.5 × more turbid than offshore areas (>24 km offshore) during the baseline (pre-dredging) period. Benthic light showed a reverse pattern, with the nearshore and deeper sites receiving ~6.5 × less light than offshore sites. This natural gradient was attributed to natural wind and wave resuspension events of the shallower nearshore area,
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and also to the influence of riverine discharges from the nearby Ashburton River. Dredging increased turbidity by 1.3–2.6×, with the largest change observed in the nearshore area where most dredging occurred, and reduced benthic light availability to 0.3–0.4× that of pre-dredging levels at sites closest to the excavation. Seabed particle size distributions 1-month after dredging were finer, and the relative proportion of clay and silt (<60 µm; fines) decreased with increasing distance from the shipping channel. This pattern suggests that sediments released by the dredging activities (spilled) resulted in a sediment deposition field adjacent to the channel. After dredging (1-month), seabed particle sizes were finer than before dredging at sites as far away as 1.5 km from the channel (the furthest distance examined), but surveys conducted 7-months after dredging showed some return to the coarser pre-dredging levels. Cyclone Olywn, which passed very close to the area as a category 3 cyclone between the post-dredging PSD surveys, may have influenced the recovery process, redistributing the finer sediments over a larger area.

The seabed benthos was composed of mobile and sessile bottom-dwelling biota, and was sparse and very variable. There were occasional patches of high abundance and diversity especially associated with three-dimensional habitat in the form of reefs and isolated shoals. In the pre-dredging survey 333 specimens were collected and a further 258 specimens collected in the post-dredging survey. Out of the 168 species and operational taxonomic units (OTUs) catalogued post-dredging, 69 (42%) were new discoveries (i.e. not collected in the pre-dredging survey), comprising 48 sponges, 21 soft corals (including gorgonians), 4 hard corals, 6 ascidians and 1 bryozoan. The surveys resulted in an increase in recorded sponge species richness for the greater Pilbara region from 1,164 to 1,233 (~6% increase), and from 406 to 485 (~19% increase) for the IMCRA Pilbara Nearshore bioregion. The retrieval of >50% new sponge records for the Onslow region as a result of the 2 surveys, suggests the study area has historically been under sampled. The recovery of 150 sponge species shows that the study area supports moderately high sponge diversity. A comparatively low occurrence of phototrophic sponge species (sponges symbiotic with cyanobacteria or zooxanthellae), suggests species at the study area were adapted to naturally high levels of turbidity. All specimens collected were registered with the Western Australian Museum (WAM), and a catalogue of in situ and surface (on-deck) photographs was developed as a resource for future marine environmental studies in the area3.

Interpretation of the changes to the benthic communities before and after the dredging are complicated by 3 natural drivers which may have influenced benthos over the study period and which could have had varied effects on the different zones examined. These events include: (1) flooding of the Ashburton River, which occurred on several occasions and is likely to have influenced some of the nearshore sites with sediment and low salinity seawater (as inferred from satellite imagery), (2) the close passage of up to 5 cyclones that may have generated substantial swell in the region, subjecting the benthic communities to extreme levels of turbidity and strong wave action, and (3) a marine heatwave, which occurred just prior to the pre-dredging survey and which may have affected pre-dredging abundances of certain taxa5.

There was some evidence of changes in benthic abundances between the before and after surveys; however, the effect was relatively weak, with taxa specific responses including both decreases and increases in abundances. This included reductions in the number of hydrozoans and colonial ascidians, and an increase in the number of sponges, gorgonians and hard corals. Using a scoring system based on functional growth forms, 16 morphologies of sponges were recorded. Natural patterns of sponge functional morphology existed pre-dredging, with highest abundance of encrusting forms (up to 58%), under-representation of cup morphotypes (8%), and intermediate levels of erect and massive forms (11% to 38%). The composition of sponge functional morphology remained relatively stable post-dredging, which may indicate an established community adapted to living in environments characterized by high sediment loads and regular cyclone exposure. Changes in sponge abundance were more variable, with the Inner and Outer zones exhibiting

5 Lafratta A et al. (2016) Coral bleaching in turbid waters of north-western Australia. Mar. Freshw. Res. http://dx.doi.org/10.1071/MF15314
reductions (-10% to -17% respectively) while sites positioned mid-way of the shipping channel (Middle/Transition Zone) showing an increase (57%).

The colonial ascidians and hydrozoans showed the highest decrease in abundance after dredging, recording a 95% and 97% reduction respectively. Notably, both colonial ascidian and hydrozoan occurred at low numbers pre-dredging, ~4 individuals m⁻² and ~2 individuals m⁻² respectively, and can be ephemeral. Therefore, it is difficult to determine whether the reduction in their abundances was caused by the dredging-related turbidity, as opposed to effects of cyclones or other environmental factors, and is limited by the coarse temporal and spatial resolution available in the present study. Interestingly, hard corals showed a slight increase in abundance between surveys. Corals in the area experienced severe mortality from the bleaching event in the summer of 2011, and were found bleached again in the 2013 pre-dredging survey⁶. The small increase in the number of corals per m² over the dredging period may indicate a recovery of hard coral populations following these high mortality events.

We are unable to determine whether the increases in sponges, gorgonians and hard corals would have been greater if the dredging and the cyclones had not occurred. Nevertheless, the absence of noticeable reductions in sponge, gorgonian and hard coral cover in an area subjected to an approximate doubling of turbidity over the 2 years of dredging program, including episodic intermittent peaks in turbidity, is notable.

Water quality was managed by the proponent during the dredging campaign using a comprehensive environmental management plan which contained a zonation scheme². The area around a small shoal (End of Channel Shoal, ENDCH) located 700 m from the most seaward extent of the navigation channel, was of particular interest as there was less restrictions for water quality management required at the site. Additionally, the filter feeder communities there were least likely to be adapted to naturally high turbidity levels associated with the nearshore environment. ENDCH experienced short term acute turbidity events when dredging occurred nearby, as well as low level chronic elevations associated with westerly drift of sediment from the dredge material placement site. The relative change in water quality at the sites was one of the most pronounced, showing the highest increase of turbidity above baseline levels (2.6× pre-dredging levels) and clear reductions in light availability (0.4× pre-dredging levels). Detailed analyses of benthic communities and sponge functional morphologies around the ENDCH site (300 m to 1.5 km away), showed similar changes as the coarser scale analyses for the Inner, Middle and Outer zones.

Macroalgae exhibited the highest reduction in abundance (-61%), followed by rhodoliths (-27%). These reductions could be due to either poor water quality associated with the dredging (i.e low light levels), or to the the effects of cyclone Olwyn (March 2015), or to seasonal differences, as the before and after surveys were performed at different times of year (March and July respectively). Considering the known ephemeral nature of macroalgae, it was difficult to conclude further about the cause of the changes⁶.

⁶ Towards the end of this study additional information on the benthic cover of macroalgae, filter feeders and seagrasses became available from the compliance monitoring programs of the dredging proponent. Surveys by the proponent were conducted in December 2012 (before dredging) and December 2014 (after dredging) which was just before the passing of cyclone Olwyn (March 2015). This information included data collected at sites considered to be largely unaffected by dredging (i.e. Bessieres Island) and sites clearly affected by dredging (the Outer Channel Zone). The data has been examined to see if there are any further insights into the temporal changes in the abundance of filter feeder and macroalgae – see Appendix 1. Briefly, no significant change was detected for the filter feeders before and after dredging – which supports the stability of this group as reported in the main body of this study. Macroalgae showed a significant reduction in cover at both the Reference site and Outer Channel Zone. These changes couldn’t have been due to cyclone Olwyn (as the sites were last surveyed 4 months beforehand), and since the before and after surveys were conducted in December of each year, this reduces (but does not eliminate) the possibility that the changes were seasonal. Of particular note, is that a number of other cyclones passed near the study area between December 2012 and 2014, including Tropical Cyclone Mitchell, Narelle, Rusty and Christine. These could have resulted in the reduction of macroalgal biomass through associated swells, and particularly at Bessieres Island which is more exposed than the Outer Channel Zone, which is protected by Thevenard Island to the north. Since the compliance surveys were effectively influenced by 4 cyclones, the sampling resolution was reasonably coarse (surveys separated by two years), the physical setting of the sites were different, and the seasonality and interannual variability of abundance in macroalgae exist (see Appendix 1), we cannot confidently attribute the changes in macroalgae to any specific cause (cyclones, dredging or seasonality).
Considerations for predicting and managing the impacts of dredging

Although before-after comparisons showed some taxa-specific decreases and increases in abundance, there was no clear effect on benthic filter feeding communities that could be attributed to dredging. As emphasised in the Executive Summary, interpretation of these changes has been made difficult because of several natural disturbance events which are likely to have affected the communities both before dredging (a marine heat wave and several cyclones), during dredging (flooding of the Ashburton River and a category 4 cyclone), and after dredging (a category 3 cyclone). In future studies, any evidence of natural disturbances during the baseline and/or operational phases of a monitoring project should prompt further detailed investigations of the scale and magnitude of the disturbance. In addition, the inclusion of several reference sites, unaffected by dredging, will be useful for differentiating the effects of natural from dredging disturbances. This will assist in the interpretation of the data collected for adaptive or compliance monitoring purposes. Coral bleaching events have now been observed in many long-term dredging projects in WA7,8,9 and elsewhere in Australia10, and the world11. The likelihood of a marine heatwave (and subsequent coral bleaching event) occurring during the baseline and operational phases of extended capital dredging projects has reached a point where explicit consideration needs to be given in the dredge management plan as to how to manage the project should one occur. Similarly, for maintenance dredging activities, consideration should be given to avoiding dredging during periods where bleaching could occur (i.e. summer months), or else to have pre-agreed management responses (e.g. including reduction or cessation of turbidity generating activities) based on the scale and magnitude of the event.

According to EPA (2016)12 “… critical windows of environmental sensitivity include times of the year or particular sites where key species or ecological communities or critical processes may be particularly vulnerable to pressures from dredging …”. Avoidance of the warmest time of year, where marine heatwaves and coral bleaching events are most likely, would fall under this management philosophy — even though there is no beforehand guarantee that a temperature related disturbance will occur.

Pre-development surveys

Accounting for natural turbidity generating processes (cyclones, storms and riverine discharge)

Shallow tropical benthic communities in cyclone-prone areas are likely to experience conditions of extreme turbidity (>100 NTU over several days) due to strong wind and wave/swell resuspension. In the 4 y duration of the water quality monitoring associated with this dredging campaign, 5 tropical cyclones (TC) passed the study site (nearest gales: TC Iggy = 150+ km away; TC Lua = 17 km; TC Mitchell = 40 km; TC Rusty = 140 km and TC Christine = <10 km), with a 6th, Cyclone Olwyn, passing directly over the study area when dredging had just been completed. Heavy rainfall and coastal flooding is a prominent feature of cyclones and storms, and as with heatwaves and bleaching events, managing projects after cyclones should be addressed a priori in the dredge management plans to assist interpretation of the data collected for adaptive or compliance monitoring purposes.

The previous history of disturbance events such as floods, marine heatwaves, and cyclones and storms (number, size, proximity etc.) on the receiving environment should be considered as part of the baseline habitat description phase of a project. Reference sites which are least likely to be affected by dredging should be identified and monitored prior to and during the development phase to better differentiate between impacts from natural phenomena and from dredging. For cyclones, damage zone models exist that can predict whether a sea state is sufficient to severely damage reef communities based on wind speed, duration and

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7 Pluto LNG Development, Burrup Peninsula: WA Environmental Protection Authority Bulletin 1259, Ministerial Statement No. 757
8 Cape Lambert B project: WA Environmental Protection Authority Bulletin 1357, Ministerial Statement 840
9 Gorgon Gas Development Barrow Island Nature Reserve: WA Environmental Protection Authority Bulletin 1221 Ministerial Statement No. 800
10 Magnetic Island (Great Barrier Reef region near Townsville, Queensland): Jones RJ (2008) Coral bleaching, bleaching-induced mortality, and the adaptive significance of the bleaching response. Mar Biol 154:65-80
11 Port of Miami, Florida: Miller MW et al. (2016) Detecting sedimentation impacts to coral reefs resulting from dredging the Port of Miami, Florida USA. PeerJ Preprints
fetch, and that can be used to identify spatial patterns in historic cyclone exposure to explain habitat condition trajectories\textsuperscript{12}.

In addition, natural inclement weather such as during the wet seasons or high winds, could also translate to pulses of increased turbidity for the area through riverine flood discharge of the Ashburton River, and re-suspension of epi-benthic layer of fine sediment. Onshore-offshore gradients of turbidity and epibenthic fine sediments were recorded for the Onslow study area, whereby inshore environments closer to the mouth of the Ashburton River experiencing higher turbidity and epibenthic fines compared to more offshore sites. The relative stability of sponge, gorgonian and hard coral communities at Onslow through natural turbidity generating events and periods of dredging suggests historic periods of natural turbidity experienced by communities pre-dredging may have selected for sediment tolerant species and morphologies. Arguably, communities that occur in naturally low turbidity environments, such as at offshore coral reefs, and not having experienced historic periods of turbidity stresses, may respond differently to elevated turbidity from dredging. Consequently, it is instructive to evaluate natural turbidity and sediment characteristics of habitats proposed for future dredging, and recognise that environmental filtering may play a role in selecting for sediment tolerant taxa and/or morphologies pre-dredging, thus reducing vulnerability of these taxa to dredging related sediment. Pre-dredging data for 2 years prior to dredging was useful in understanding the natural turbidity events of the study area. To comprehensively sample and survey for changes in benthic communities, water quality data (in particular from a year of pre-dredging water quality monitoring) should be made available and critically evaluated prior to conducting the first baseline assessments. This will provide an evidence base and rationale to select reliable reference sites not/least affected by natural sources of sediment.

**Baseline assessment of benthic filter feeder communities**

From this study a 166-page colour photograph catalogue of all specimens (n = 591) collected during the two surveys was created\textsuperscript{3}. This catalogue includes in-situ and on deck photographs of sponges, hard corals, soft corals, gorgonians, ascidians, hydrozoans and bryozoans for the Onslow area. Individual functional morphologies scored for the sponges and used in this study were also included. Importantly, all of these specimens are now registered with the Western Australian Museum, and can be referred to for any future taxonomic work. Proponents of future survey and monitoring programs at or around the area are encouraged to use this field guide for standardisation of data with respect to sponge functional morphology and benthic taxa species identification where relevant.

The reliability and suitability of survey and assessment methods depends on the characteristics of benthic communities, and should be specifically tailored to maximise accuracy of the data gathered. Benthic communities at the Onslow study area were patchily distributed; therefore, sparse and small bodied taxa may be easily overlooked when examining photographic images and using a 5-point intercept technique which is commonly used in long-term monitoring programs using underwater images. To maximise accuracy, this study counted all individuals present in each towed video image, which required a high degree of effort to complete. In addition, a single assessor was used to score benthic taxa to eliminate observer bias. Alternatively, the number of point intercepts may be increased to improve data accuracy. As such, we recommend power analyses to be performed on a subset of towed video images to assess the optimal number of points (e.g. 20, 50 etc.) to be used in accurately capturing the abundance of sparse and patchily distributed benthic taxa.

**Impact prediction**

To reliably predict the impact from dredging, temporal and spatial water quality data collected during baseline and dredging should be assessed using appropriate statistical methods, to characterise the temporal variability typically encountered during dredging operations (e.g. periodic increases in turbidity as dredging occur in proximity to water monitoring sensors). In addition, the occurrences of natural turbidity generating events such as cyclones, which can influence turbidity and benthic taxa inside (impact) and outside (reference) of dredging

\textsuperscript{12} Puotinen M et al. (2016) A robust operational model for predicting where tropical cyclone waves damage coral reefs. Sci Rep 6:26009
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zones should also be considered; to allow more reliable inference of the effects of dredging on benthic communities and to exclude any confounding effects of natural phenomena.

**Appropriate statistics for evaluating long-term water quality**

Capital dredging operations may take up to several years to complete as opposed to maintenance dredging which generally take several weeks. Even within extended capital dredging projects, sensitive water quality receptors may only be exposed to turbidity plumes for durations of a few weeks as dredges pass through the area – for example when dredging along an entrance channel. However, this is not always the case, for example, dredging in the nearshore areas for turning basins may be more protracted and require larger volumes to be removed on account of shallower water depth at the coastline. Nevertheless, an important question is over what period of time should water quality data (i.e. NTUs and light availability) be examined to characterize the effects of dredging as compared to background natural turbidity events. The water quality analyses showed that receptors were exposed to episodic peaks in turbidity (i.e. acute effects) as well as more extended (chronic) elevations in turbidity (see also). To characterize this temporal variability, this study assessed water quality over multiple time periods (1, 7, 14 and 30 d running mean intervals), for the 50th percentile values ($P_{50}$) as well as the $P_{80}$, $P_{95}$ and $P_{100}$ for turbidity (NTU), and $P_{20}$, $P_{50}$, $P_{95}$ and $P_{100}$ for daily light integrals (DLI). The use of running mean percentile statistics account for acute pulses (e.g. turbidity: $P_{95}$ at 1 d running mean interval) and chronic levels of stressors (e.g. $P_{80}$ at 30 d running mean interval) over the entire dredging phase. Therefore, predictive modelling for the assessment of dredging impacts would benefit from using the more informative running mean percentile statistics as opposed to mean and median statistics which overlook these temporal patterns; as different benthic taxa may respond differently to acute or chronic stresses.

**Effects of cyclones and natural disturbances**

Cyclones can profoundly affect water quality over large spatial scales resulting in extreme levels of turbidity. In this study, several cyclones occurred during the baseline water quality monitoring resulting in periods of extreme turbidity levels. This can affect summary statistics and interpretation of the data if the aim is to compare multiple short-term acute and longer term chronic effects of dredging with the baseline phase. If reference and dredging affected sites are separated over small spatial scales, and are under the influence of the same natural phenomena, then these sorts of extreme events could be removed from water quality datasets, on a case-by-case basis for future modelling efforts.

**Monitoring**

**Water quality assessments**

Comprehensive water quality data, for turbidity and light, and benthic particle size distribution (PSD) were made available by the dredging proponent for analysis in this study. This dataset included 2 years of baseline water quality data and 2 years of data during the dredging phase, as well as the dredge logs, which showed production rates, and when and where dredging was occurring. These data were analysed using the same techniques (broad summary statistics, exceedance curves and analyses such as running means percentile analysis from 1 d to 30 d) as used with water quality data from other similarly sized dredging projects. This has enabled analyses of the intensity, duration and frequency of turbidity events during the baseline and dredging phases, as well as an analysis of temporal and spatial effects. The summary tables derived from these analyses have provided a very useful, practical resource and a reference point for contextualising any effects on water quality for future dredging projects. The analyses have also provided a useful resource for laboratory-based studies on filter feeders (examining cause-effect pathways and dose-response relationships), allowing testing to be conducted using environmentally realistic and relevant exposure scenarios hence contextualising the results of the

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13 Ports Australia (2014) Dredging and Australian Ports. Subtropical and Tropical Ports. Ports Australia, Sydney, NSW Australia: 96 pp.
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Dredging proponents should be encouraged to make all fully QA/QC’d data from water quality investigations available for future analysis to enable this resource to grow.

Using sensitive benthic filter feeder and associated taxa to detect dredging-related stress

Most sponge species are generally long-lived and abundance can be stable over time. However, seasonal variations for other filter feeder taxa have been reported in other global regions, for example in the Mediterranean, where seasonal dormancy of colonial ascidians and hydrozoans may result in lowered detectable visual abundance (see\(^{20}\) for review). Similarly, seasonal dormancy has also been reported in macroalgae, a taxa which co-occurred in filter feeder habitats\(^{21}\). In this study, there was a clear reduction in macroalgae (-61%), colonial ascidian (-94%) and hydrozoan (-97%) abundances after dredging, which may be a response to reduced light penetration (reduction in photosynthetically active radiation to the benthos for macroalgae) and sensitivities to increases in suspended sediment concentrations (affecting feeding and gas exchange of colonial ascidians and hydrozoans). Declines may also be attributable to the occurrence of cyclones and flooding of the Ashburton River. Surveys performed at different times of year (March vs July, summer vs winter, etc.), can add further complexities to interpretations on the effects of dredging on filter feeder communities, as seasons have been reported to influence the abundance and distributions of benthic taxa\(^{20}\). To eliminate any confounding effects of seasonal variation on benthic community abundances, surveys should be conducted during the same time of year where possible, with reliable reference sites (unaffected by dredging stressors) monitored concurrently. Additionally, more frequent surveys (e.g. every 2-3 months) would be highly beneficial in assessing the natural variability of ephemeral species.

This is the first study to effectively use the functional morphology concept for sponges\(^{22}\) for a dredging scenario. The results showed under-representation of cup morphotypes even before-dredging. Notably, up to 80% of sponges found on some reefs on the clear water Great Barrier Reef were reported to be cup or foliose shaped, and coincide with their phototrophic mode of energetic acquisition\(^{23,24}\). This suggests that the naturally turbid environment, caused by riverine discharge of the Ashburton River, could have limited the occurrence and persistence of cup and photosynthetic sponges at Onslow. Interestingly, a study on turbid reefs situated at the mouth of the largest riverine system in the world, the Amazon River, reported similar sponge morphological composition\(^{25}\) (Onslow/Amazon; Encrusting = 20%–58%/ 21%, Massive = 11%–29%/ 27%, Erect = 15%–38%/ 25%, Cup = 3%–8%/ 10%). Based on the morphological structure of Onslow and the Amazon River mouth sponges, the proportion of cup morphotypes at ≤~10% may indicate sponge populations that are well adapted to living in turbid environments and that are less sensitive to dredging-related stressors. On the other hand, sponge populations having high abundances of cup morphotypes, for example 50 – 80% on “clean-water” reefs of the Great Barrier Reef or at similar environments of NW Western Australia, may not be as tolerant to elevated turbidity and light attenuation associated to dredging.

Increased turbidity (lower water clarity) corresponds to reductions in light penetration through the water column, thus reducing the amount of photosynthetically active radiation to the benthos. The sensitivity of cup

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16 Pineda et al. (2016) Effects of light attenuation on the sponge holobiont-implications for dredging management. Scientific Reports. 6:39038
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and phototrophic sponges to light attenuation were further corroborated by controlled laboratory studies in Theme 6\textsuperscript{16}. Specifically, bleaching (i.e. discolouration from loss of photosymbionts) occurred within 7 days in the phototrophic sponge *Carteriospongia foliascens* when exposed to experimental darkness (0 mol photons m\textsuperscript{-2} d\textsuperscript{-1}). After 28 d of exposure in low light conditions (<0.8 mol photons m\textsuperscript{-2} d\textsuperscript{-1}), *C. foliascens* was not able to recover from bleaching with high mortality incurred. **Cup and phototrophic species such as Carteriospongia spp. (sensitive species), and its bleaching response with light attenuation, represent suitable visual indicators for the monitoring and management of dredging related stressors.** When used in conjunction with water monitoring programs, early detection of discolouration in phototrophic sponges between reference and impact zones (to account for natural variation) can inform dredging proponents of any detrimental stress to phototrophic sponge populations, which allows for immediate preventative measures to be taken.

**Management**

The management of dredging operations, based on the framework of the EPA (2016) guideline\textsuperscript{2}, requires the prediction of spatially-explicit zones: the Zones of Influence, Moderate Impact, and High Impact. **Data gathered in this field study, on water quality parameters of turbidity, light attenuation and epibenthic sediment dynamics during a large-scale and long-term capital dredging program, coupled with data on thresholds levels (including LC\textsubscript{10} and LC\textsubscript{50} for these stressors on a number of sponge species encompassing differing modes of feeding and functional morphologies (see Theme 6 – Project 4 for a summary table), provides a platform for better informed modelling (i.e. sediment transport and ecological response) of these zonal boundaries.** Using phototrophic cup sponges as examples, their low representation at the naturally turbid Onslow environment concurs with results of controlled laboratory experiments which identified the sensitivity of this particular functional morphology to high suspended sediment concentrations (i.e. turbidity) and low light availability. Through the modelling of dredging plumes (an objective of Theme 2 and 3), Zones of High Impact (i.e. high SSCs and low light; e.g. LC\textsubscript{50}), Moderate Impact (i.e. moderate SSCs and moderate light; e.g. LC\textsubscript{10}) and Influence (i.e. low SSCs and high light) can be identified. Sentinel receptors, such as *in situ* nephelometers and light loggers, can be then be strategically positioned at these modelled boundaries to detect trajectories of exceedance.

**Residual Knowledge Gaps**

This project has improved our understanding of changes in water quality patterns from dredging, and its effects on benthic filter feeder communities at the site of Western Australia’s largest capital dredging program to date. In addition, it also identified the study area off Onslow to be a diverse habitat for sponges with 150 species recorded. This study also highlighted a number of knowledge gaps; of which additional research investment would contribute further to managing the impacts of dredging on filter feeder communities in north west Western Australia.

**Reproductive biology and population dynamics**

Global information on reproductive biology of sponges, such as modes of sexuality and development, fecundity, seasonality and spawning periods are in general depauperate. At present, information on the reproductive biology of northwestern tropical Australian sponges is limited to three bioeroding species\textsuperscript{26}. Notably, more information on sponge reproduction is available for southwestern temperate species across more

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\textsuperscript{26} Fromont J et al. (2005) Excavating sponges that are destructive to farmed pearl oysters in Western and Northern Australia. *Aquaculture Research*. 36:150-162
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morphologies\textsuperscript{27,28,29}, likely facilitated by easier access to study sites in proximity to metropolitan Perth. Similarly, information on larval ecology are equally lacking with data on larval dispersal, settlement rates and recruitment success limited to a handful of Australian Great Barrier Reef sponge species, and no work conducted on Western Australian sponges to date. Considering sponges are important components of filter feeder communities, and represent an important biodiversity resource for NW WA (i.e. 1233 species and OTUs of sponges from the Pilbara alone\textsuperscript{30}), it is important to understand processes contributing to population maintenance and recovery. Periods when populations are reproductive and spawning represent critical phases of the lifecycle of sponges, with larval and juvenile phases likely to be more sensitive to external pressures than adults. Understanding when sponges are reproductive and spawning, and appreciating the sensitivities of larvae to dredging related pressures would assist in identifying environmental windows for dredging (i.e. time of the year when key species, ecological communities or critical processes may particularly be vulnerable to dredging pressures). Importantly, local ecological and environmental knowledge on the reproduction, larval dispersal, settlement behaviours and recruitment patterns are critical in setting effective environmental windows for the management of dredging-related pressures. In addition, understanding the levels of connectivity between populations and processes of replenishment through larval dispersal and supply from outside of affected areas, and understanding processes of recruitment and post-settlement survival, would allow for predictions on community population recovery following dredging.

**Taxonomic resolution**

Northwestern Australia is a global hotspot for sponges. At present 1233 species and OTUs of sponges have been recorded for the Pilbara region, of which only 15% are formally described\textsuperscript{30}. The effects of dredging stressors on sponges are species-specific. For example, phototrophic cup sponges have been shown to be sensitive to high turbidity and low light environment, conditions of which are associated to dredging. Therefore, it is important to accurately identify sponge species and their distributions when assessing the effects of dredging. For sponges, taxonomic identification is especially difficult with gross morphologies providing insufficient information for species identification, thus requiring microscopic assessments of spicules and spongine fibres. Additionally, morphological plasticity exists in sponges and adds further complexities for species delineation, and sometimes require molecular phylogenetic techniques for resolving taxonomy. There is an overwhelming backlog of OTUs requiring taxonomic identification, progress of which would give added confidence for future monitoring of sponge communities to the effects of dredging. Similarly, taxonomic resolution for other filter feeder groups, such as ascidians and bryozoans are equally poor, and highlights the need for taxonomic expertise for these groups in Western Australia.

**Sponge associated fauna**

Sponges can add significant spatial complexity and can serve as habitats to other fauna. Despite the high biodiversity and abundance of sponges in tropical northwest Western Australia, little is known of the diversity, abundance and distribution of sponge associated fauna, with most of these associations potentially cryptic (i.e. living in and on the sponge). Of relevance to assessing the effects of dredging on sponge communities, the presence or absence of endofauna (those living in the sponge) and/or epifauna (those living on the sponge) may serve as a indicators for sponges to dredging related stresses. Therefore, establishing baseline data on sponge-associated fauna associations, particularly for a group of commonly occurring sponge species across the Pilbara, may contribute to the development of early warning protocols for sponge responses to dredging related stresses.

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A R T I C L E  I N F O

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A B S T R A C T

Changes in turbidity, sedimentation and light over a two year large scale capital dredging program at Onslow, northwestern Australia, were quantified to assess their effects on filter feeder communities, in particular sponges. Community functional morphological composition was quantified using towed video surveys, while dive surveys allowed for assessments of species composition and chlorophyll content. Onslow is relatively diverse recording 150 sponge species. The area was naturally turbid (1.1 mean P90 NTU), with inshore sites recording 6.5 × higher turbidity than offshore localities, likely influenced by the Ashburton River discharge. Turbidity and sedimentation increased by up to 146% and 240% through dredging respectively, with corresponding decreases in light levels. The effects of dredging were variable, and despite existing caveats (i.e. bleaching event and passing of a cyclone), the persistence of sponges and the absence of a pronounced response post-dredging suggest environmental filtering or passive adaptation acquired pre-dredging may have benefited these communities.

1. Introduction

Changes in water quality can influence the abundance, diversity and structure of benthic communities in freshwater (Burdon et al., 2013; Milner et al., 2016), estuarine (Jones, 1996; Tweedley et al., 2012), and marine ecosystems (Fabricius, 2005; Knapp et al., 2013; Stubler et al., 2015). Elevated suspended sediment concentrations (SSCs) in particular can have a range of effects on tropical marine communities (Rogers, 1990; Fabricius, 2005; Erftemeijer et al., 2012; Bell et al., 2015; Jones et al., 2016). Firstly, high SSCs and associated turbidity (water cloudiness) can attenuate light for photosynthesis by autotrophic taxa, and subsequent light limitation can reduce energy for growth and reproduction in seagrasses and scleractinian corals (Anthony and Hoegh-Guldberg, 2003; Fabricius, 2005; Collier et al., 2012; Jones et al., 2016). In addition, the finer fractions of suspended solids can clog feeding and respiratory apparatus of filter feeders such as sponges (Tompkins-MacDonald and Leys, 2008), ascidians (Armsworthy et al., 2001), bivalves (Ellis et al., 2002) and barnacles (Fabricius and Wolanski, 2000). Furthermore, the settlement of suspended material results in increased sedimentation, which can affect communities through smothering or the need for energetically demanding processes such as self-cleaning (Edmunds and Davies, 1989; Stafford-Smith and Ormond, 1992; Riegl and Branch, 1995; Bannister et al., 2012). Lastly, suspended sediments can also negatively influence reproductive output and life cycles of sessile benthic invertebrates such as scleractinian corals and sponges (Whalan et al., 2007; Jones et al., 2015b; Ricardo et al., 2016).

Anthropogenic activities such as dredging can temporarily result in very high SSCs (Jones et al., 2015a; Fisher et al., 2015), and are usually subjected to environmental impact assessment (EIA) processes. Within the EIA framework, the potential effects on sessile benthic primary producers such as seagrasses, mangroves, seaweeds, and scleractinian corals are often emphasized, as these are habitat-forming taxa supporting a myriad of other marine species such as fish and invertebrates (EPA, 2011). Of the habitat-forming taxa, comparatively little is known on how sponges respond to changes in suspended and settled sediment (Bell et al., 2015). Although the literature clearly shows that sponges are influenced by sediment in a variety of ways, most studies confer that...
sponges are able to tolerate, and in some cases thrive, in environments subject to sedimentation (Bell et al., 2015; Moura et al., 2016).

Concerns over the effects of elevated sediments on sponge communities have come to the fore-front in recent years in the Pilbara region of north-west Western Australia where macrobenthic filter feeder grounds can dominate the benthos at many locations (Heyward et al., 2010; Przeslawski et al., 2014). The region has also been the subject of intense industrial activities in the last decade associated with a resource boom, and there have been many large-scale capital dredging projects for port expansions and creation of shipping channels to new coastal liquefied natural gas (LNG) facilities (Hanley, 2011). In a recent study of sponge biodiversity in the Pilbara, 1164 Linnaean species and operational taxonomic units (OTUs) of sponges were recorded (Fromont et al., 2016), far surpassing the diversity of scleractinian corals on the Great Barrier Reef (n = 405, see Devantier et al., 2006). Despite the diversity and functional importance of sponges in the region, information on their biology, ecology and responses to turbidity and sedimentation pressures are comparatively limited. This is a challenge to the effective management of this biodiversity resource (EPA, 2011; Bell, 2008).

The sensitivity of sponges to sedimentation is known to vary between taxa with different morphologies, with individuals possessing encrusting and tube morphology shown to be more sensitive to sedimentation than erect forms (de Voogd and Cleary, 2007). Phototrophic (chlorophyll containing) sponges are common in shallow tropical reef environments (Wilkinson, 1983, 1988), and the long term light attenuation that occurs during dredging projects (see Jones et al., 2016) could lead to loss of symbionts and symbiont-containing species (Thacker, 2005; Roberts et al., 2006). The differential response of sponges which have various morphologies and levels of phototrophy could potentially be used as indicators for the management of dredging related stresses and pressures on filter feeder communities.

In this study, we examine the response of filter feeder and sponge communities to a very large scale (~31.4 Mm$^3$) and extended (~2 year) capital dredging project near Onslow in the Pilbara region of north-west Western Australia (WA; Wheatstone Project; see Ministerial Approval Statement MS 873, available on the WA EPA website: http://www.epa.wa.gov.au). The environmental regulation and permitting process associated with this project were comprehensive, requiring the relocation of ~31.4 Mm$^3$ of sediment (Ministerial Approval Statement MS 873; Fig. 1). The dredging involved construction of a 16 km shipping channel 12 km south-west of Onslow, and dredging and armouring the coastal portion of a gas trunckline west of the channel, and required the relocation of ~31.4 Mm$^3$ of sediment to several dredge material placement sites, the largest of which was located 9 km northeast of the end of the shipping channel (Fig. 1).

Dredging was undertaken using trailer suction hopper dredges, cutter suction dredges and backhoe dredges and occurred near continuously (24 h a day) and sometimes with multiple dredges working concurrently in different locations.

The study area was 10 km west of the town of Onslow and 8 km east of the Ashburton River mouth (Fig. 1). The area has an arid tropical climate with often extended periods of drought, and with the majority of rain falling between January and June. The area experiences land-sea breeze cycles, with winds that are overall southerly to westerly in spring and summer months, and that are less defined in the winter months. The area is under the influence of cyclones (with an average of 5 in WA each year), typically passing through the area in southerly or south-easterly tracks. The Ashburton River can have a significant local coastal influence with highly variable conditions from zero to intense periods of flow generally associated with cyclones. In the nearshore area, tides are semi-diurnal with a moderate tidal range of 1.9 m for spring tides. Local topography directs the tidal currents along the coast in an easterly direction on the flood tide and westerly flow on the ebb tide, although flow patterns can be interrupted by wind-driven currents in particular during neap tides. Tidal currents are moderate and in the coastal area are not significantly influenced by large scale ocean current systems. The coastal area is dominated by partially lithified and unconsolidated alluvial sediments partially overlayed by sediments of marine origin. Net alongshore sediment transport is generally from west to east based on the wave climate and prevailing winds (Chevron, 2010).

2.2. Water quality

Turbidity and light were measured using turbidimeters (ECO-NTU-SB OBS turbidity recorder) and light loggers (ECO-PAR-SB) at 30 min intervals for ~2 y before dredging (17 May 2011–10 April 2013) and for ~2 y during dredging (11 April 2013–27 February 2015) at 16 locations positioned throughout the study area at depths of 6–12 m (see Fig. 1 and Table 1). Turbidity and light loggers were deployed vertically 1.3 m and 1.5 m above the seabed using landers. Pre-dredging, data were downloaded every 6 weeks. During dredging, the loggers were telemetered and data were downloaded every 30 min. Nephelometric turbidity unit (NTU) and photosynthetically active radiation (PAR) data were plotted and checked for quality and ambiguous data points removed using the approach outlined in Jones et al. (2015a). To allow visual comparisons of long term (chronic) turbidity patterns, the 80th, 95th, 99th and 100th (maximum) percentiles ($P_{80}$, $P_{95}$, $P_{99}$ and $P_{100}$) of NTU were calculated for running mean intervals between 1 h and 30 d. A Generalised Additive Model (GAM) was used to estimate PAR values (400–750 nm) for every second throughout the daylight period following Jones et al. (2015a). The sum of the per second quantum flux measurements were then added to calculate the daily light integral (DLI) as mol photons m$^{-2}$ d$^{-1}$. The 20th, 10th, 5th, 1st and 0th (minimum) percentiles ($P_{20}$, $P_{10}$, $P_{5}$, $P_{1}$ and $P_{0}$) of the DLI were calculated for running mean intervals between 1 h and 30 d. Percentiles were calculated using the runmean and runquantile functions from the caTools package in R (Tuszyński, 2013; R Core Team, 2014). Additionally, NTU and DLI cumulative probability curves were developed to assess exceedence and reduction of dredging NTUs and DLIs respectively from pre-dredging levels. A conversion factor for NTU to SSC (mg L$^{-1}$) ranged between 1.6 and 2.6 (mean = 2.2) for the Wheatstone project (Peter Fears pers. comms.).

2.3. Particle size distribution of benthic sediment

Grab samples for particle size distribution (PSD) analyses were collected along the shipping channel in December 2012 (prior to the start of dredging), immediately after dredging (March 2015) and in October (2015) ~7 months later. Duplicate samples for superficial sediment were collected using a 2.4 L Petite Pona® (Wildco, FL, US) grab at sampling stations 100, 250, 750 and 1500 m perpendicular to the channel along 7 transects, centred on the channel and at increasing...
Table 1

| Site no. | Site name       | Abbreviation | Latitude      | Longitude     | Depth (m) |
|----------|-----------------|--------------|---------------|---------------|-----------|
| 1        | Bessiers Island | BESS         | 21.53022      | 114.77045     | 7.5       |
| 2        | Roller Shoal    | ROLLER       | 21.6495       | 114.926003    | 9.3       |
| 3        | Ashburton Island| ASHNE        | 21.58923      | 114.940033    | 9.3       |
| 4        | Paroo Shoal     | PAROO        | 21.5648       | 115.079002    | 10.2      |
| 5        | Saladin Shoal   | SALAD        | 21.57032      | 115.03055     | 9.7       |
| 6        | Thevenard Island| THIE         | 21.4834       | 115.025717    | 9.1       |
| 7        | Thevenard Island| THIE         | 21.45112      | 115.034167    | 10.2      |
| 8        | End of channel  | ENDCI        | 21.53413      | 115.052633    | 11.2      |
| 9        | Gorgon Patch    | GORGSW       | 21.355        | 115.069033    | 9.8       |
| 10       | Ward Reef       | WARD         | 21.6092       | 115.079002    | 8.3       |
| 11       | Weeks Shoal     | WEEKS        | 21.52038      | 115.093617    | 11.3      |
| 12       | Direction Island| DRINE        | 21.52693      | 115.135417    | 7.3       |
| 13       | Twin Islands    | TWIN         | 21.50748      | 115.1955      | 6.7       |
| 14       | Herald Reef     | HERALD       | 21.48653      | 115.215233    | 7.2       |
| 15       | Airlie Island   | AIREL        | 21.32817      | 115.1569      | 6.3       |
| 16       | West Reef       | WEST         | 21.32648      | 115.3913      | 9.6       |

Fig. 1. Atmospherically and colour corrected, pan sharpened Landsat image from the United States Geological Survey (USGS) Operational Land Imager (OLI) instrument of the study area on 24th August 2013 during the dredging phase, showing the location of (1) the study site in the Pilbara Region of NW Australia (see Inset A) 12 km west of the coastal township of Onslow and east of the Ashburton River mouth. A sediment plume from the Ashburton River (see Ashburton River mouth) can be seen migrating eastwards close to the shore, (2) the entrance channel, gas trunkline (dashed line east of the channel) and dredge placement site (spoil ground; yellow box), (3) the 16 water quality monitoring sites (refer to Table 1 for details). Sites are coloured according to pre-dredging nephelometer turbidity unit (NTU) clusters (light blue = cluster 1, yellow = cluster 2, orange = cluster 3 and red = cluster 4; see Fig. 2), (4) towed video transect mid-points (blue diamonds) and dive survey sites (red stars) partitioned into Inner (green dashed box), Mid (red dashed box) and Outer channel zones (blue dashed box; see Inset B) and (5) the seabed particle size distribution (PSD) sampling sites (S1–S7; green dots). See Supplementary Fig. 5 for time series Landsat images of the study area reflecting the dynamic nature of sediment plumes. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.) Landsat images produced by M. Broomhall and P. Fearns, Remote Sensing and Satellite Research Group, Curtin University. Source of Landsat data: US Geological Survey.

2.4. Towed video surveys

Underwater towed video surveys were conducted to examine changes in abiotic substrate and benthic community composition along the shipping channel, partitioned into 3 zones (Inner, Middle [Mid] and Outer) based on the PSD assessment described above (Fig. 1). Although the ‘Mid’ channel zone did not contain any PSD sampling transects, it was included to assess any potential transitional changes in benthic communities along the channel between the Inner and Outer zones. Towed video surveys were conducted on 18–30 March 2013 (pre-dredging) and 3–13 July 2015 (post-dredging), along 9 transects in the Inner zone, 7 in the Mid zone and 11 in the Outer zone. Live-feed videos of the seafloor and benthos were captured onto miniDV tapes by towing a video body over the survey area, and using a variable-speed winch to control height of the video above the sea bed. The length of the towed video transects ranged from 350 to 780 m (median = 390 m) with the number of photos available for analyses ranging from 45 to 109 per transect (Supplementary Table 1). Depth of individual towed video transects were determined from the vessel’s depth sounder. High resolution still images (with a field of view between 0.3 and 0.5 m) were captured at 10 s intervals (which corresponded to ~10 m spatial interval) using a 12 MP digital still camera mounted to the towed video body. All devices (computer, video recorder and stills cameras) were sub-second synchronised to GPS (latitude, longitude) with date and time stamps. For further details of these remote imaging systems see Speare et al. (2008) and Negri et al. (2010).

Images from the towed video surveys were examined by a single assessor to eliminate observer bias. Abiotic substrate composition was processed using the Australian Institute of Marine Science (AIMS) Long Term Monitoring protocol (see Sweatman et al., 2008), and the proportion of the benthos occupied by silt, sand, unconsolidated pebble/gravel, shells, rubble and reefal substrate quantified. As biota were
sparse in the study area, assessment at a higher resolution was performed which involved counting all individuals within each image. A total of 18 biological categories (i.e. macroalgae, rhodolith, seagrass etc. — see Table 4) were used. For colonial taxa, a collective of individuals was considered as a single entity based on relative colony size (10 individuals for colonial ascidians, 5 for zooanthids and 5 for hydrozoans). For cnidarians, such as hard corals, soft corals and gorgonians, which form colonies but have more distinct gross morphologies, a single colony is one which is clearly separated from another. Sponges were further categorised into 16 functional morphological groups (i.e. encrusting, erect, massive etc. — see Table 6) using categories defined by Schönberg and Fromont (2014). The total number of biota recorded was divided by total area surveyed (assuming each image is 0.5 × 0.5 m) to attain a standardised measure of abundance (individual m⁻²). While the image FOV ranged from 0.3–0.5 m, the larger image area was used to attain a conservative measure of abundance, relevant for the sparsely distributed benthic communities encountered in the study.

2.5. Sponge species assessments

Dive surveys (n = 12) were conducted on 18–30 March 2013 (pre-dredging) and 3–13 July 2015 (post-dredging), to assess fine scale, species level changes in sponge communities (Fig. 1). At each site 2–4 transects (1 × 5 m) were laid out haphazardly and surveyed for sponges, ascidians, bryozoans and cnidarians (including gorgonians, soft corals and hydrozoans). The dive sites between pre- and post-dredging surveys were at the same location, however, permanent transects were not established. Numbers of individuals were recorded for sponge species and operational taxonomic units (OTUs), and representative specimens photographed in situ and then sampled. Sub-samples were preserved in liquid nitrogen for chlorophyll a (Chl-a) analysis and transferred to −80 °C for storage. Remaining sub-samples were preserved in 75% ethanol for taxonomic work. Sponge abundance was averaged by area of replicate transects for each dive site (individuals 5 m⁻²), and individual dives considered as replicates for comparison of species composition between pre- and post-dredging surveys. Species diversity indices (Simpson diversity index (1 − λ′) and Pielou’s evenness index (J′)) and total number of species (S) were calculated based on species occurrences and abundance. Average quantitative taxonomic diversity (Δ) and distinctness (Δ′), which consider taxonomic relatedness of species, were also calculated.

2.6. Sponge chlorophyll a analyses

Sponge Chl-a concentration was used to infer phototrophic status pre- and post-dredging. Frozen samples were left to thaw for 15 min and placed in dry, pre-weighed 15 mL extraction tubes and wet weight (ww; mean = 360 mg) determined using an analytical balance. Laboratory grade methanol (99.6%) was added to the tubes to immerse – 3.8 g of ww sponge) of known photosynthetic sponges as reported by Cheshire et al. (1997) and Wilkinson (1983). Sponges with a Chl-a concentration of < 2.6 μg g⁻¹ ww sponge were not considered to be phototrophic, based on the heterotrophic sponge Ianthella basta (Cheshire et al., 1997). Sponges having Chl-a concentrations > 32.9 μg g⁻¹ ww sponge, were considered as having moderate phototrophy, based on the partially autotrophic species Jaspis stellifera. Sponges with a Chl-a concentration of > 63.5 μg g⁻¹ ww sponge were considered to have a high photosynthetic capacity, based on phototrophic cyanosponge species such as Carteriospongia folialis and Phyllospongia papayracea (Bannister et al., 2011; Webster et al., 2012).

2.7. Statistical analyses

All univariate and multivariate analyses were performed in PRIMER and PERMANOVA+, using 9999 permutations of the resemblance matrix where relevant (V7, Clarke and Gorley, 2015). Where required, ANOSIM test was performed to assess main effects of independent variables and PERMANOVA performed to assess interactions in multivariate assessments. Cluster analysis of pre-dredging NTU P_so data across all running mean intervals was conducted to aid identification of sites exhibiting similar natural patterns of turbidity (Euclidean distance matrix). The P_so was selected for analyses as it preserves useful information, including intermittent turbidity peaks from natural and dredging events, for assessing chronic effects of NTU on benthic communities which would otherwise be masked using mean and median statistics (Fisher et al., 2015; Jones et al., 2015a).

To assess differences in PSĐ between dredging periods and at distances away from the shore, two-way ANOSIM and PERMANOVA tests were used (Bray-Curtis similarity matrix). Pairwise comparisons were further conducted to identify where differences exist. Metric MDS (mMDS without zero-intercept; 100 restarts, minimum stress = 0.001) of the data averaged within transects (grabs away from channel as replicates) was performed to visualise differences in PSĐ between dredging periods. Vector correlation analysis (Pearson’s correlation) was conducted to identify variables (PSĐ size classes) contributing most to the pattern observed in the MDS ordination space and SIMPROF test performed to identify statistically distinct groups. The effects of dredging and distances away from channel were assessed using the same method using grabs away from shore as replicates within transects.

One-way PERMANOVA was used to assess differences in sampling depth between Inner, Mid and Outer channel zones (Euclidean distance matrix). To assess differences in superficial substrate composition between dredging periods (pre- and post-dredging) and channel zones, two-way ANOSIM and PERMANOVA were used (Bray-Curtis similarity matrix). Pairwise comparisons were further performed to detect where specific differences exist. ANOSIM and PERMANOVA tests were similarly used to detect differences in benthic community composition and sponge functional morphologies between dredging periods and channel zones. A univariate two-way PERMANOVA test was additionally used to assess changes in individual filter feeder taxa (i.e. sponge, gorgonians, ascidians and hydrozoans) and photautotrophs (i.e. macroalgae) between dredge periods and channel zones. RELATE and BEST tests were performed to assess any relationships between benthic community structure and substrate composition, and to identify the most relevant substrate types influencing benthic community structure. To assess effects of the highest dredging impact, specific tests were additionally performed on benthic community composition and sponge functional morphologies data from transects closest to the site showing highest increase in turbidity (n = 6; End of Channel).

One-way ANOSIM was used to assess differences in sponge species composition between pre- and post-dredging (Bray-Curtis similarity matrix). SIMPER analysis was used to identify species contributing most to differences between dredging periods (Bray-Curtis similarity matrix, 70% cut-off). One-way univariate PERMANOVA on total number of species (S), average quantitative taxonomic diversity (Δ) and
distinctness ($\Delta^*$), using dive sites as replicates, were performed to assess changes in species richness, diversity and evenness pre- and post-dredging (Euclidean distance matrix).

3. Results

3.1. Water quality

Turbidity was highly variable across all 16 water monitoring sites and prominent features include (1) transient and episodic peaks in turbidity at sites close to the dredging as compared to more distantly located control sites, and (2) abrupt transient peaks in turbidity (with maximum turbidity exceeding > 100 NTU) over a short period when tropical cyclones passed through the study area (Fig. 2A and Supplementary Fig. 1). Daily light integrals showed an inverse pattern where light level decreased when turbidity peaked. In addition, periods of low DLI s occurred during the passing of clouds, with deeper sites (i.e. End of Channel and Weeks) displaying lower DLI values compared to adjacent shallower sites (Fig. 2A and Supplementary Fig. 1). Since the cyclones had broad effects over the entire study area, including control and dredge sites — and since no dredging was conducted during these periods — the data points associated with these events were removed (refer to Supplementary Table 2 for information on cyclones). Assessment of NTU percentiles ($P_{90}$, $P_{95}$, $P_{99}$ and $P_{100}$) of different running mean periods (from 1 h to 30 d) showed similar patterns, with $P_{100}$ decreasing as temporal scale was increased from hours to weeks (Fig. 2A and Supplementary Fig. 1). Values for $P_{90}$ and $P_{95}$ were relatively stable across shorter time intervals (hours to 7 d) but showed a decrease at > 7 d intervals, and values for $P_{99}$ were stable across all time scales examined. Cumulative probability curves of NTU showed an increase in turbidity during dredging at all sites, with the NTU exceeding $P_{99}$, median = 1.88, range = 0–8.7. Baseline: mean = 1.94, median = 0.98, range = 0–27.2. Reduction = 191.

Cluster analysis of $P_{99}$ turbidity values across running mean time intervals showed broad effects over the entire study area, including control and dredge sites — and since no dredging was conducted during these periods — the data points associated with these events were removed (refer to Supplementary Table 2 for information on cyclones). Assessment of NTU percentiles ($P_{90}$, $P_{95}$, $P_{99}$ and $P_{100}$) of different running mean periods (from 1 h to 30 d) showed similar patterns, with $P_{100}$ decreasing as temporal scale was increased from hours to weeks (Fig. 2A and Supplementary Fig. 1). Values for $P_{90}$ and $P_{95}$ were relatively stable across shorter time intervals (hours to 7 d) but showed a decrease at > 7 d intervals, and values for $P_{99}$ were stable across all time scales examined. Cumulative probability curves of NTU showed an increase in turbidity during dredging at all sites, with the NTU exceeding $P_{99}$, median = 1.88, range = 0–8.7. Baseline: mean = 1.94, median = 0.98, range = 0–27.2. Reduction = 191.

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Table 2
Summary of turbidity (NTU) percentile values (P50, P80, P95, and P100) for running mean intervals of 1, 7, 14 and 30 d for the 16 water monitoring sites for the Wheatstone dredging project. Pre-dredging NTU clusters refer to sites having similar NTUs over the two year pre-dredging period, with sites within cluster 1 having the lowest turbidity to cluster 4 having the highest turbidity. The ratio of dredging/baseline NTUs at the 80th percentile (P80) is provided in the last column. Cell colours represent relative strength of NTU and dredging/baseline ratio values, with dark green representing low values and dark red representing high values.

| Site  | Pre-dredging NTU | Cluster | Before 1 d | Before 7 d | Before 14 d | Before 30 d | During 1 d | During 7 d | During 14 d | During 30 d | Ratio change P80 |
|-------|------------------|---------|------------|------------|-------------|-------------|------------|------------|-------------|-------------|-----------------|
| BESS  | 0.3 0.4 0.8 7.4  | 0.3 0.4 0.6 1.1 | 0.3 0.4 0.6 0.7 | 0.3 0.4 0.5 0.6 | 0.6 0.8 1.1 9.1 | 0.7 0.8 1.1 3.9 | 0.7 0.8 1.0 2.4 | 0.6 0.8 0.9 1.5 | 1.9 1.9 2.0 2.1 |
| THIESE| 0.5 1.3 2.0 5.1  | 1.0 1.2 1.6 2.6 | 1.0 1.1 1.4 2.5 | 0.9 1.0 1.2 1.3 | 1.7 2.2 3.0 6.3 | 1.7 2.2 2.8 3.8 | 1.6 2.1 2.6 3.5 | 1.6 1.9 3.0 3.1 | 1.7 1.9 1.9 1.9 |
| ASHNE | 0.7 1.0 1.6 6.9  | 0.7 1.0 1.7 2.3 | 0.6 1.1 1.5 1.8 | 0.7 1.1 1.2 1.2 | 1.3 1.7 2.7 15.8 | 1.3 1.8 2.6 6.8 | 1.3 1.8 2.7 4.4 | 1.3 1.6 2.1 2.8 | 1.7 1.9 1.7 1.5 |
| AURIE | 0.6 1.2 1.9 5.4  | 0.7 1.2 1.6 3.2 | 0.7 1.2 1.4 2.9 | 0.9 1.1 1.6 1.9 | 1.1 1.7 2.8 59.0 | 1.2 1.7 2.3 48.0 | 1.2 1.6 2.3 32.0 | 1.2 1.6 2.1 22.0 | 1.4 1.4 1.4 1.5 |
| THIE  | 0.6 0.9 1.2 3.2  | 0.7 0.9 1.2 1.5 | 0.7 0.8 1.1 1.3 | 0.6 0.8 1.0 1.0 | 1.3 1.9 3.0 65.0 | 1.4 1.9 2.6 40.0 | 1.3 1.9 2.5 34.0 | 1.3 1.9 2.7 30.0 | 2.1 2.2 2.3 2.3 |
| WEEKS | 1.0 1.4 2.1 7.4  | 1.0 1.3 1.9 5.9 | 1.1 1.3 1.8 3.8 | 1.1 1.2 1.4 2.1 | 1.7 2.4 40.2 57.0 | 1.8 2.5 3.9 10.4 | 1.8 2.5 3.3 6.4 | 1.7 2.2 2.4 2.3 | 1.7 1.9 1.8 1.8 |
| PAROO | 0.7 1.0 1.7 6.2  | 0.7 1.0 1.7 2.7 | 0.7 1.2 1.7 1.8 | 1.0 1.3 1.5 1.5 | 1.4 2.2 44.1 16.7 | 1.6 2.4 4.3 6.1 | 1.6 2.4 3.9 4.9 | 1.6 2.4 3.2 3.7 | 2.3 2.3 2.0 1.8 |
| ENDC  | 0.9 1.3 1.9 5.2  | 1.0 1.3 1.8 2.1 | 1.0 1.3 1.6 1.8 | 0.9 1.3 1.4 1.5 | 2.2 3.3 57.0 24.0 | 2.4 3.3 50.0 10.1 | 2.3 2.2 4.4 8.5 | 2.2 2.7 4.1 6.0 | 2.5 2.6 2.4 2.1 |
| SALAD | 0.8 1.2 1.9 5.5  | 0.9 1.2 1.9 2.4 | 0.9 1.2 1.7 1.9 | 0.9 1.2 1.4 1.4 | 1.7 2.9 66.0 30.6 | 1.8 3.1 6.1 11.7 | 1.9 3.0 5.5 7.3 | 2.0 2.8 5.1 5.5 | 2.3 2.5 2.5 2.4 |
| HERALD| 0.8 1.4 2.5 10.0 | 0.9 1.5 2.2 3.6 | 1.0 1.6 2.1 2.4 | 1.1 1.5 1.8 2.2 | 1.4 2.2 42.0 34.9 | 1.5 2.2 3.7 12.8 | 1.6 2.2 2.9 7.5 | 1.7 2.1 2.6 4.6 | 1.6 1.5 1.4 1.4 |
| TWIN  | 1.0 1.7 2.9 9.0  | 1.1 1.7 2.4 4.4 | 1.2 1.8 2.3 2.8 | 1.3 1.6 2.1 2.6 | 1.7 2.6 51.0 41.2 | 1.8 2.7 4.4 16.6 | 1.9 2.7 3.6 9.8 | 2.1 2.6 3.4 3.4 | 1.5 1.6 1.5 1.6 |
| GORGSW| 1.0 1.5 2.3 11.8 | 1.0 1.5 2.2 4.0 | 1.2 1.4 2.1 2.6 | 1.2 1.5 1.7 1.8 | 1.7 2.7 55.0 31.5 | 1.9 3.0 5.4 9.2 | 2.1 2.9 4.6 7.1 | 2.1 3.0 4.1 4.8 | 1.9 2.0 2.0 2.0 |
| WEST  | 0.8 1.9 3.5 22.7 | 0.9 2.0 2.9 11.1 | 1.1 2.1 2.5 6.4 | 1.2 2.0 2.2 4.2 | 1.0 1.6 31.2 12.4 | 1.1 1.7 3.0 5.6 | 1.2 1.8 2.4 3.4 | 1.2 1.7 2.2 2.5 | 0.9 0.9 0.9 0.8 |
| DIRNE | 1.2 1.9 3.3 13.7 | 1.3 2.0 2.9 4.9 | 1.4 2.0 2.7 3.1 | 1.6 2.0 2.5 2.8 | 1.6 2.5 44.0 50.9 | 1.7 2.6 40.0 21.1 | 1.8 2.7 3.7 13.0 | 1.9 2.5 3.3 7.7 | 1.3 1.3 1.3 1.3 |
| WARD | 1.3 2.3 4.1 14.1 | 1.4 2.5 3.9 6.5 | 1.7 2.7 3.2 4.6 | 1.5 2.6 3.1 3.8 | 2.3 4.2 97.0 46.9 | 2.5 4.5 8.9 23.9 | 2.6 4.8 7.9 14.5 | 2.5 4.4 5.5 9.1 | 1.8 1.8 1.8 1.7 |
| ROLLER| 1.7 2.6 4.4 17.5 | 1.8 2.5 3.7 5.5 | 1.8 2.6 3.3 3.6 | 2.1 2.6 2.8 2.8 | 2.4 4.8 11.9 43.4 | 2.8 5.4 10.0 19.4 | 2.9 6.0 8.8 12.5 | 3.4 6.3 8.2 9.1 | 1.9 2.1 2.3 2.4 |
intervals found 4 distinct clusters at 1.9 Euclidean distance similarity (Supplementary Fig. 2). Plots of pre-dredging $P_{60}$ NTUs at multiple running mean intervals showed consistent patterns within groups and differences between groups (no overlap; Fig. 2B). During dredging, increases in $P_{60}$ NTU were variable within the pre-dredging NTU clusters with different dredging/before dredging ratios at 1 d running mean interval ranging from 0.9–2.5 (Fig. 2B and Table 2). For example, within NTU cluster 2, while pre-dredging NTU were similar, sites further from dredging (e.g. AIRLIE and ASHNE) showed lower increase in NTU compared to sites that were closer to dredging activities (e.g. ENDCH and SALAD, Fig. 2B and Table 2). Before dredging, $P_{20}$ DLI values ranged between 1.3 and 8.7 (at running mean interval of 1 d), with the inshore ROLLER (DLI 1.3), and deeper ENDCH (1.4) and WEEKS (1.4) recording lower light levels than shallow, offshore sites (e.g. BESS, 8.7 and TWIN 4.1; Supplementary Table 3).

The highest increase of turbidity above baseline levels was at the End of Channel (ENDCH) water quality monitoring site, located ~700 m from the dredge channel. The site experienced both short term acute elevations in turbidity (> 50 NTU from the 30 min readings, and a mean maximum 1 d turbidity of 25 NTU) when dredging occurred close by, as well as more low level, chronic increases in turbidity associated with westerly drift of sediment from the dredge material placement sites (see Figs. 1, 2, and Supplementary Fig. 5). The $P_{60}$ turbidity over 1, 7, 14 and 30 d running mean periods was 2.1–2.6 x higher than the baseline, which was the highest ratio recorded for all water quality monitoring sites (Table 2). The ENDCH site also showed one of the highest decreases in light levels during dredging, with $P_{20}$ DLI over 1, 7, 14 and 30 d running mean periods ranging between 0.3 and 0.4 x of baseline levels.

3.2. Particle size distribution of benthic sediment

Metric MDS (mMDS) of averaged PSD showed clustering of sites pre-dredging with only the most inshore site (S1) separating from the rest of the sampling sites (Fig. 3A). SIMPROF test recovered statistically distinct groups with the offshore sites S5 and S6 showing least change in PSD composition across all dredging periods. Vector analyses showed that separation between pre- and post-dredging PSD was correlated (Pearson’s correlation $r > 0.9$) to increases in clay (< 2 µm) and silt fractions (2–60 µm), with the increase in combined clay and silt fractions ranging from 120 to 240% at inshore sites compared to ~7 to 103% at offshore sites (Fig. 3A; Table 3). Seven months after the dredging, PSDs were distinct from post-dredging patterns with some inshore sites (S2, S3 and S4) resembling the S1 site pre-dredging and was correlated to increase in the smaller sand fraction (60–150 µm). No significant change was detected in PSDs at site S1 1 month after dredging and 7 months after dredging as determined by SIMPROF (Fig. 3A). ANOSIM analyses showed a significant effect of dredging period and distance from shore on PSDs (Sig. level = 0.01% for both factors); however, these relationships were relatively weak (R = 0.244 and 0.217 respectively). Pairwise comparisons showed the largest effect strength between pre- and post-dredging surveys (R = 0.300), and between inshore (S1, S2, S3 and S4) and offshore sites (S5, S6 and S7; maximum R = 0.627). PERMANOVA analysis showed no significant interaction between factors ($p > 0.05$).

The mean proportion of fine fractions (clay and silt combined), was consistent across distances away from the edge of the shipping channel before dredging, ranging between 19 and 22%, and was reflected in clustering of sites in the mMDS ordination space (Fig. 3B). One month after dredging, the mean proportion of fines (range: 39–55%) approximately doubled, with this proportion decreasing with increasing distances away from the channel i.e. 54% (100 m away from the channel), 55% (250 m away), 44% (750 m away) and 39% (1500 m away; Fig. 3B). In the survey conducted 7 months after the dredging, there was up to a 46% reduction in proportion of fines (Fig. 3B; Table 3). Two-way PERMANOVA showed a significant interaction between dredging period (pre-dredging, 1 and 7 months after dredging) and distance away from the channel for PSD (df = 6, pseudo-F = 2.14, $p = 0.02$). One way PERMANOVA for individual survey periods showed significant effects of distance away from the channel for PSDs 1 and 7 months after dredging ($p < 0.05$), but this was not significant before dredging.

3.3. Visual census of superficial abiotic substrate

Mean sampling depths were significantly different between zones and were deepest at the Outer zone (range: 11.9–14.7 m), followed by the Mid zone (9.6–10.8 m) and Inner zone (7.9–10.7 m; PERMANOVA: df = 2, pseudo-F = 96.643, $p < 0.001$; see Supplementary Table 1 for detailed towed video transect information). Silt and sand (combined) formed the majority of the substrate, constituting between 86 and 98% across time periods and zones (Fig. 4). Before dredging, the Outer zone exhibited the highest proportion of silt (78 ± 4%) followed by the Inner (61 ± 3%) and Mid zones (21 ± 2%). The Mid channel zone showed the highest increase in silt (67 ± 2%; 211% increase) immediately after dredging followed by the Inner zone (90 ± 2%; 48% increase), with the Outer zone exhibiting least change in silt composition (80 ± 2%; 2% increase). Two-way ANOSIM showed significant effects of time period and zone on superficial substrate composition, with a relatively high effect strength for both factors (Sig.
level = 0.01%, R = 0.565 and 0.535 respectively). PERMANOVA analysis revealed a significant interaction between dredging period and channel zones (df = 2, pseudo-F = 10.455, p = 0.0001). Pairwise comparisons showed no significant difference in substrate composition between the Inner and Outer channel zones (PERMANOVA: $t = 1.46$, $p = 0.11$).

### 3.4. Benthic community composition

Sessile taxa dominated the benthos across dredge periods (pre- and post-dredging) and channel zones (Inner, Mid and Outer), with motile taxa including Asteroidea, Echinoidea, Holothuroidea, Nudibranchia and Polychaeta forming < 0.1 individual m$^{-2}$ combined (< 2% of total counts; Fig. 5). Macroalgae, rhodolith and hard corals were most dominant in the Mid channel zone before dredging. Macroalgae showed the highest reduction in abundance post-dredging (−61%), followed by rhodoliths (−27%; Fig. 5, Table 4). Filter and suspension feeders, including sponges, ascidians, gorgonians and hydrozoans were dominant at the Inner and Outer channel zones before dredging. The responses of filter feeders to dredging was variable, with the most consistent pattern detected in gorgonians which showed an increase in abundance ranging from 13 to 90% before and after dredging and across all zones (Table 4). A decrease in sponge abundances was detected for the Inner and Outer zones (−16% and −10% respectively), with an increase in the Mid channel zone of 57%. Colonial ascidians, which were the most dominant taxa in the Outer channel zone, showed the highest reduction in abundance post-dredging (−94%; Table 4).

ANOSIM test showed significant effects of dredging periods and channel zones on benthic community composition; however, the effect strength is relatively weak (Significance = 0.01%; R = 0.24 and 0.28 respectively). PERMANOVA identified no significant interaction of dredging period and channel zone (df = 2, pseudo-F = 1.833, $p = 0.076$). Pairwise comparison of channel zones showed no significant difference in substrate composition between the Inner and Outer channel zones (PERMANOVA: $t = 1.46$, $p = 0.11$).

### Table 3

Summary of mean (± SE) PSD proportions (%) along the 16 km shipping channel during pre- and post-dredging, and at 10 months from end of dredging. Transects S1 to S7 run from inshore to offshore consecutively (see Fig. 1 for map). Values for proportions of sand and gravel are means of sums of all size classes within the particle type group.

| Transect | Dredge period | % clay (< 2 μm) Mean SE | % silt (2-60 μm) Mean SE | % sand (60-2360 μm) Mean SE | % gravel (> 2360 μm) Mean SE |
|----------|----------------|-------------------------|--------------------------|-----------------------------|-----------------------------|
| 1        | Pre            | 21.00 1.80              | 14.86 2.12               | 59.86 3.69                  | 4.29 0.76                   |
|          | Post           | 32.81 2.40              | 45.91 3.23               | 19.62 4.33                  | 1.66 0.85                   |
|          | Recovery       | 28.00 1.72              | 30.00 3.64               | 41.05 4.88                  | 0.95 0.45                   |
| 2        | Pre            | 12.88 0.93              | 5.75 0.92                | 62.13 3.72                  | 19.25 4.35                  |
|          | Post           | 24.27 2.98              | 39.26 4.31               | 29.24 4.77                  | 7.23 2.53                   |
|          | Recovery       | 18.63 3.84              | 15.00 4.72               | 60.47 6.78                  | 5.91 2.52                   |
| 3        | Pre            | 13.25 2.85              | 9.88 3.08                | 59.88 4.49                  | 17.00 4.01                  |
|          | Post           | 25.78 1.99              | 34.46 2.69               | 31.53 3.26                  | 8.23 2.68                   |
|          | Recovery       | 21.50 3.08              | 15.69 2.76               | 49.64 2.32                  | 13.17 4.05                  |
| 4        | Pre            | 9.88 1.88               | 12.13 2.12               | 51.38 7.69                  | 26.63 8.02                  |
|          | Post           | 22.86 2.60              | 34.69 4.84               | 31.48 5.64                  | 10.97 4.94                  |
|          | Recovery       | 22.56 3.64              | 18.75 4.22               | 51.30 7.23                  | 7.38 3.10                   |
| 5        | Pre            | 8.13 0.97               | 4.88 0.95                | 62.25 7.82                  | 24.75 8.13                  |
|          | Post           | 11.89 1.76              | 13.45 1.72               | 47.60 8.92                  | 27.05 11.36                 |
|          | Recovery       | 8.19 2.63               | 4.31 1.13                | 69.13 4.56                  | 18.37 5.32                  |
| 6        | Pre            | 11.13 0.52              | 9.00 1.48                | 61.38 6.42                  | 18.50 5.53                  |
|          | Post           | 8.11 1.21               | 10.55 1.64               | 64.67 6.06                  | 16.66 4.66                  |
|          | Recovery       | 10.00 1.37              | 6.63 0.91                | 70.77 3.67                  | 12.60 3.42                  |
| 7        | Pre            | 9.88 1.17               | 6.38 1.02                | 57.75 4.67                  | 26.00 5.54                  |
|          | Post           | 13.42 1.20              | 19.58 1.89               | 45.62 3.52                  | 21.39 3.99                  |
|          | Recovery       | 18.00 5.07              | 10.94 3.54               | 50.76 6.45                  | 20.30 5.03                  |

Fig. 4. Proportion composition of superficial substrate assessed visually from images collected through towed video surveys. A, B and C represent substrate composition pre-dredging, and D, E and F represent substrate composition post-dredging. A and D represent Outer channel zone, B and E Mid channel zone and C and F Inner channel zone.
and a significant main effect of dredging periods only on hydrozoan abundance ($p = 0.0001$ and 0.108). Significant main and interaction effects of dredging period and channel zones were detected for macroalgae ($p = 0.015$).

RELATE tests showed a significant correlation between abundances of benthic communities and superficial substrate composition both before and after dredging (Significance levels = 0.01% and 0.09% respectively); however, this relationship was stronger pre-dredging (Spearman’s $Rho = 0.448$) than post-dredging (Spearman’s $Rho = 0.250$). The BEST test identified silt, unconsolidated pebble/gravel, shells and reefal substrate to be most influential in shaping benthic communities pre-dredging, with silt having the strongest effect (Table 5). Only silt and pebble/gravel unconsolidated substrates were significantly correlated to benthic community composition post-dredging.

Specific tests on data from transects closest to the site of highest impact (ENDCH; mid-points ranging from 0 to 680 m from the edge of channel; $n = 6$) showed no multivariate effects of dredging on the benthic community (PERMANOVA $p = 0.109$), however univariate tests on individual taxa showed significant effects on hydrozoans (negative effect, PERMANOVA $p = 0.014$) and hard coral (positive effect, PERMANOVA $p = 0.028$), and to a certain extent colonial ascidians (negative effect, PERMANOVA $p = 0.051$; Supplementary Figs. 3 and 4).

3.6. Sponge species assessments

Before dredging, 333 specimens were collected, and after identifications in situ and further taxonomic work in the laboratory, 102 sponge, 20 cnidarian (hard and soft corals, and gorgonians), 18 ascidian, 6 bryozoan and 3 hydrozoan Linnaean species and OTUs were identified. After the dredging, a further 258 specimens were collected and from these 90 sponge, 55 cnidarian, 18 ascidian, 4 bryozoan and 1 hydrozoan Linnaean species and OTUs were identified. Of the 168 species and OTUs identified in the post-dredging surveys, 69 (42%) had not been collected before dredging, and comprised 48 sponges, 21 soft corals, 4 hard corals, 6 ascidians and 1 bryozoan. Of the 150 total sponge Linnaean species and OTUs recovered, 69 (46%) were new records for the Pilbara region, 10 (7%) had been found previously in the Pilbara but not for the Pilbara Nearshore bioregion of Australia’s Integrated Marine and Coastal Regionalisation spatial classification framework (IMCRA, 1998, 2006), and 71 (47%) were known previously from this bioregion (see Supplementary Table 4). Recovery of sponge species in this study increased species richness for the greater Pilbara region to 1233, and to 485 for the IMCRA Pilbara Nearshore bioregion.

One-way ANOSIM showed a significant difference in species composition between pre- and post-dredging periods (Significance = 0.02%, $R = 0.667$). High between-group dissimilarity was detected for sponge species composition pre- and post-dredging, with 49 species contributing 71% to this dissimilarity (SIMPER average dissimilarity = 90%; Table 7). Of these 49 species, 18 occurred only at either the pre- or post-dredging survey. Notably, 4 species which contributed 11% of dissimilarity between groups occurred at high abundances, but were found only at one of the sampling periods (i.e. pre- or
Table 4
Summary table of mean densities (± SE; individual m⁻²) of benthic taxa at Onslow sampled pre- and post-dredging, and at the Inner, Mid and Outer channel zones. n represents the number of replicate towed video surveys at each sampling period and channel zone. Ratio change corresponds to density post-dredging/pre-dredging, where a value of 1 = no change, 0 = complete absence of individuals post-dredging and values > 1 representing an increase in individual taxa numbers (red text represents a decrease and blue text represents an increase in density post-dredging). Asterisk (*) highlights where individuals were only found post-dredging and not pre-dredging. γ represents taxa which showed reductions in density post-dredging at all zones across the channel and β represents taxa which showed increases in density post-dredging at all zones across the channel.

| Taxa            | Inner Pre (n = 9) | Inner Post (n = 9) | Ratio change | Mid Pre (n = 7) | Mid Post (n = 7) | Ratio change | Outer Pre (n = 11) | Outer Post (n = 11) | Ratio change |
|-----------------|-------------------|-------------------|--------------|-----------------|-----------------|--------------|-------------------|-------------------|--------------|
| Macroalgae      | 2.469 ± 0.670     | 0.097 ± 0.037     | 0.039        | 5.090 ± 3.429   | 3.526 ± 0.786   | 0.388        | 0.186 ± 0.073     | 0.072 ± 0.021     | 0.388        |
| Rhodolith       | 0.108 ± 0.039     | 0.059 ± 0.039     | 0.547        | 4.803 ± 2.341   | 3.490 ± 0.877   | 0.727        | 0.381 ± 0.195     | 1.390 ± 0.744     | 3.651        |
| Seagrass        | 0.033 ± 0.026     | 0.021 ± 0.021     | 0.636        | 0.012 ± 0.012   | 0.154 ± 0.115   | 13.201       | 0.028 ± 0.019     | 0.014 ± 0.014     | 0.495        |
| Hard coral      | 0.740 ± 0.215     | 0.809 ± 0.442     | 1.094        | 1.558 ± 0.486   | 3.011 ± 0.705   | 1.932        | 0.569 ± 0.123     | 1.028 ± 0.215     | 1.805        |
| Soft coral      | 0.046 ± 0.016     | 0.043 ± 0.038     | 0.920        | 0.045 ± 0.022   | 0.133 ± 0.023   | 2.984        | 0.053 ± 0.022     | 0.029 ± 0.011     | 0.550        |
| Gorgonian       | 0.934 ± 0.292     | 1.504 ± 0.783     | 1.610        | 0.281 ± 0.150   | 0.319 ± 0.082   | 1.137        | 0.752 ± 0.183     | 1.429 ± 0.452     | 1.901        |
| Zoanthid        | 0.004 ± 0.004     | 0.039 ± 0.021     | 9.662        | 0.045 ± 0.035   | 0.000 ± 0.000   | 0.000        | 0.037 ± 0.017     | 0.016 ± 0.016     | 0.424        |
| Sponges         | 2.357 ± 0.611     | 1.963 ± 0.751     | 0.833        | 0.822 ± 0.410   | 1.290 ± 0.503   | 1.570        | 2.722 ± 0.641     | 2.452 ± 0.080     | 0.901        |
| Ascidian solitary | 0.068 ± 0.042   | 0.022 ± 0.012     | 0.320        | 0.000 ± 0.000   | 0.087 ± 0.058   | *           | 0.012 ± 0.008     | 0.022 ± 0.012     | 1.773        |
| Ascidian colonial | 0.750 ± 0.220   | 0.684 ± 0.408     | 0.912        | 0.123 ± 0.076   | 0.130 ± 0.083   | 1.052        | 4.531 ± 1.983     | 0.256 ± 0.086     | 0.057        |
| Hydrozoan       | 2.192 ± 0.304     | 0.669 ± 0.445     | 0.632        | 0.207 ± 0.077   | 0.443 ± 0.413   | 2.137        | 0.771 ± 0.133     | 0.190 ± 0.153     | 0.234        |
| Bryozoan        | 0.100 ± 0.043     | 0.088 ± 0.032     | 0.887        | 0.030 ± 0.014   | 0.075 ± 0.031   | 2.491        | 0.315 ± 0.109     | 0.270 ± 0.113     | 0.858        |
| Asteridea       | 0.011 ± 0.008     | 0.009 ± 0.006     | 0.801        | 0.010 ± 0.010   | 0.000 ± 0.000   | 0.000        | 0.028 ± 0.015     | 0.037 ± 0.032     | 1.335        |
| Crinoida        | 0.000 ± 0.000     | 0.015 ± 0.007     | *           | 0.000 ± 0.000   | 0.000 ± 0.000   | 1.000        | 0.000 ± 0.000     | 0.000 ± 0.000     | 1.000        |
| Echiroidea      | 0.000 ± 0.000     | 0.000 ± 0.000     | 1.000        | 0.030 ± 0.019   | 0.000 ± 0.000   | 0.000        | 0.047 ± 0.036     | 0.000 ± 0.000     | 0.000        |
| Holothuroidea   | 0.020 ± 0.020     | 0.004 ± 0.004     | 0.218        | 0.000 ± 0.000   | 0.000 ± 0.000   | 1.000        | 0.007 ± 0.007     | 0.000 ± 0.000     | 0.000        |
| Nudibranchia    | 0.000 ± 0.000     | 0.000 ± 0.000     | 1.000        | 0.000 ± 0.000   | 0.000 ± 0.000   | 1.000        | 0.006 ± 0.006     | 0.007 ± 0.007     | 1.269        |
| Polychaeta      | 0.000 ± 0.000     | 0.000 ± 0.000     | 1.000        | 0.000 ± 0.000   | 0.000 ± 0.000   | 1.000        | 0.000 ± 0.000     | 0.050 ± 0.043     | *            |

*Means that individuals only found at post-dredging survey.
post-dredging. For example, *Raspalia keriontria* and *Pseudoceratina* sp. 2 which were found at mean abundances of 1.5 ± 0.4 and 0.9 ± 0.2 individuals m\(^{-2}\) post-dredging respectively, were absent pre-dredging, while the opposite was true for *Axinella aruensis* Type II and *Psammocinia* cf. *bulbosa* (1.1 ± 0.3 and 1 ± 0.4 individuals m\(^{-2}\) respectively). Assessment of univariate diversity indices showed that total number of species (S) was 12% higher before dredging than after dredging which translated to marginally higher species, average quantitative taxonomic diversity and distinctness, between the pre- and post-dredging surveys (PERMANOVA: p = 0.470

### Table 5

| # | Cumulative Rho | Abiotic variables |
|---|---|---|
| Pre-dredging | | |
| 1 | 0.395 | Si |
| 2 | 0.449 | Si, Pe |
| 3 | 0.472 | Si, Pe, Sh |
| 4 | 0.479 | Si, Pe, Sh, Re |
| Post-dredging | | |
| 1 | 0.214 | Si |
| 2 | 0.324 | Si, Pe |

Post-dredging). The majority of sponge species (89%) had comparatively low or no Chl-a content (Fig. 7). Four species were identified as having high Chl-a content. *Sarcotragus* cf. sp. SS7, which possessed the highest level of Chl-a (172 μg g\(^{-1}\) ww sponge, n = 1), and *Ectyoplasia frondosa* (66 ± 41 μg g\(^{-1}\) ww sponge, n = 3) were only found in the pre-dredging survey. Two species, *Hyatella* cf. *intestinalis* (79 ± 16 μg g\(^{-1}\) ww sponge, n = 2) and *Agelas* cf. *mauritiana* (74 ± 73 μg g\(^{-1}\) ww sponge, n = 2) were found in both the pre- and post-dredging surveys

3.7. Sponge chlorophyll a analyses

The Wheatstone dredging project was Western Australia’s largest single capital dredging project to date, and involved the excavation and relocation of ~31.4 Mm\(^3\) of sediment using multiple types of dredges, working sometimes simultaneously and near continuously in multiple locations over an extended (~2 year) period. Water quality was managed by the proponents using a comprehensive environmental management plan which contained a zonation scheme (see EPA, 2011) that allowed areas of high, moderate and no biological impact depending on proximity to the dredging. Sediment released (spilled) during excavation had clear effects on water column turbidity (where increased turbidity corresponded to reduced light penetration) over large spatial scales and the subsequent settling of the sediment created deposition zones around the dredging areas, increasing the silt content of the

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**Fig. 6.** Mean (± SE) pre- and post-dredging sponge functional morphology densities (individuals m\(^{-2}\)) at the A) Outer, B) Mid and C) Inner channel zone.
Table 6
Summary table of mean densities (± SE; individual m⁻³) of sponge functional morphologies at Onslow sampled pre- and post-dredging, and at the Inner, Mid and Outer channel zones. n represents number of replicate towed video surveys at each sampling period and channel zone. Ratio change corresponds to density at post-dredging/pre-dredging, where a value of 1 = no change, 0 = complete absence of individuals post-dredging and values > 1 representing an increase in individual taxa number (red text represents a decrease and blue text represents an increase in density post-dredging). Asterisk (*) highlights where individuals were only found post-dredging and not pre-dredging. β represents taxa which showed increments in density post-dredging across the channel. Abbreviation: EN = encrusting, cg = creeping, en = endolithic, cr = crust; M = massive, s = simple, bl = ball, crp = cryptic; C = cup, tab = tabulate, inc = incomplete, wd = wide, b = barrel, nr = narrow; E = erect, lam = lamellate, pal = palmate, br = branching, s = simple and st = stalked.

| Functional morphology | Inner Pre (n = 9) | Post (n = 9) | Ratio change | Inner Pre (n = 7) | Post (n = 7) | Ratio change | Inner Pre (n = 11) | Post (n = 11) | Ratio change | Ratio change |
|-----------------------|------------------|-------------|--------------|------------------|-------------|--------------|------------------|-------------|--------------|--------------|
|                       | Mean (n = 9)     | Post (n = 9) | Mean SE      | Mean SE          | Ratio change | Mean SE      | Mean SE          | Ratio change | Mean SE      | Ratio change |
| EN–cg                 | 0.237 ± 0.071    | 0.120 ± 0.077 | 0.505 ± 0.022 | 0.022 ± 0.014    | 0.083 ± 0.037 | 3.818 ± 0.022 | 0.160 ± 0.058    | 0.176 ± 0.071 | 1.100 ± 0.022 |             |
| EN–en                 | 0.076 ± 0.034    | 0.046 ± 0.025 | 0.611 ± 0.035 | 0.035 ± 0.035    | 0.148 ± 0.096 | 4.232 ± 0.035 | 0.074 ± 0.037    | 0.122 ± 0.054 | 1.641 ± 0.035 |             |
| EN–cr                 | 1.047 ± 0.211    | 0.482 ± 0.127 | 0.460 ± 0.389 | 0.210 ± 0.407    | 0.116 ± 0.164 | 1.045 ± 0.035 | 1.133 ± 0.357    | 0.443 ± 0.111 | 0.391 ± 0.035 |             |
| M–s                   | 0.257 ± 0.066    | 0.229 ± 0.071 | 0.890 ± 0.057 | 0.034 ± 0.161    | 0.086 ± 0.282 | 2.825 ± 0.034 | 0.468 ± 0.104    | 0.312 ± 0.094 | 0.666 ± 0.034 |             |
| M–bl                  | 0.050 ± 0.021    | 0.027 ± 0.015 | 0.547 ± 0.012 | 0.012 ± 0.021    | 0.014 ± 1.811 | 1.133 ± 0.029 | 0.074 ± 0.030    | 0.874 ± 0.030 |             |
| M–crpβ                | 0.244 ± 0.111    | 0.299 ± 0.191 | 1.227 ± 0.020 | 0.020 ± 0.095    | 0.043 ± 4.734 | 0.146 ± 0.058 | 0.332 ± 0.110    | 2.276 ± 0.030 |             |
| C–tab                 | 0.043 ± 0.020    | 0.020 ± 0.010 | 0.459 ± 0.024 | 0.024 ± 0.039    | 0.019 ± 1.634 | 0.031 ± 0.017 | 0.016 ± 0.016    | 0.510 ± 0.030 |             |
| C–inc                 | 0.028 ± 0.063    | 0.031 ± 0.017 | 2.273 ± 0.000 | 0.000 ± 0.017    | 0.017 ± 0.024 | 0.010 ± 0.008 | 0.008 ± 0.032    | 0.328 ± 0.032 |             |
| C–wd                  | 0.011 ± 0.007    | 0.005 ± 0.005 | 0.493 ± 0.012 | 0.012 ± 0.000    | 0.000 ± 0.000 | 0.000 ± 0.012 | 0.008 ± 0.033    | 0.016 ± 2.779 |             |
| C–brβ                 | 0.011 ± 0.007    | 0.004 ± 0.001 | 5.763 ± 0.000 | 0.000 ± 0.009    | 0.009 ± 0.000 | 0.000 ± 0.000 | 0.000 ± 0.000    | 0.000 ± 0.005 |             |
| C–nr                  | 0.000 ± 0.000    | 0.000 ± 0.000 | 1.000 ± 0.000 | 0.000 ± 0.000    | 0.000 ± 1.000 | 0.000 ± 0.000 | 0.000 ± 0.000    | 0.000 ± 0.000 |             |
| E–lam                 | 0.128 ± 0.088    | 0.277 ± 0.143 | 2.160 ± 0.150 | 0.088 ± 0.103    | 0.068 ± 0.196 | 0.063 ± 0.329 | 0.152 ± 1.673    |             |
| E–pal                 | 0.020 ± 0.014    | 0.015 ± 0.007 | 0.752 ± 0.012 | 0.012 ± 0.000    | 0.000 ± 0.000 | 0.000 ± 0.013 | 0.009 ± 0.022    | 1.630 ± 0.015 |             |
| E–brβ                 | 0.136 ± 0.065    | 0.225 ± 0.086 | 1.653 ± 0.049 | 0.032 ± 0.141    | 0.048 ± 0.241 | 0.073 ± 0.441 | 0.230 ± 1.832    |             |
| E–s                   | 0.070 ± 0.024    | 0.081 ± 0.037 | 1.156 ± 0.042 | 0.030 ± 0.067    | 0.042 ± 1.596 | 0.128 ± 0.051 | 0.126 ± 0.040    | 0.986 ± 0.040 |             |
| E–st                  | 0.000 ± 0.000    | 0.010 ± 0.010 | *             | 0.000 ± 0.000    | 0.000 ± 1.000 | 0.005 ± 0.014 | 0.010 ± 2.659    |             |

*Means that individuals only found at post-dredging survey.
surrounding seabed. Broad taxa specific effects were identified including both positive and negative responses. Likewise, the broad- and fine-scale surveys of sponges showed variable effects of turbidity and sedimentation on sponge functional morphology between locations along the shipping channel.

Table 7
Summary of sponge species contributing to > 70% of average dissimilarity between pre- and post-dredging as determined by similarity percentage (SIMPER) analysis (Average dissimilarity = 89.51). SIMPER ranks species according to the overall percentage contribution each makes to the average dissimilarity between groups. Average density within pre- and post-dredging reflects square root of transformed density values.

| Species                      | Pre-dredging avg. density | Post-dredging avg. density | Avg. diss. SD diss. | Individual % contribution | Cumulative % contribution |
|------------------------------|---------------------------|----------------------------|---------------------|---------------------------|----------------------------|
| Raspailia kerorientia<sup>a</sup> | 0                         | 1                          | 2.93                | 1.26                      | 3.27                       |
| Axinella arunsi Type II<sup>b</sup> | 0.85                      | 0                          | 2.41                | 1.31                      | 2.69                       |
| Pseudoceratina sp. 2<sup>a</sup> | 0.8                       | 0.8                        | 2.37                | 1.44                      | 2.65                       |
| Psammocinia bullosa cf.<sup>a</sup> | 0.77                      | 0                          | 2.31                | 0.87                      | 2.58                       |
| Axos flabelliformis           | 0.57                      | 0.74                       | 2.15                | 1.23                      | 2.4                      |
| Oceanapia sp. 7 cf.           | 0.7                       | 0.09                       | 2.03                | 0.91                      | 2.27                       |
| Rensociapha stalinagmus       | 0.74                      | 0.09                       | 1.92                | 1.17                      | 2.15                       |
| Ceratopogon axiforum          | 0.5                       | 0.55                       | 1.92                | 0.99                      | 2.15                       |
| Ianthea flabelliformis        | 0.6                       | 0.55                       | 1.89                | 1.1                       | 2.11                       |
| Mycale mirabilis              | 0.54                      | 0.08                       | 1.8                 | 0.95                      | 2.01                       |
| Oceanapia sp. 5               | 0.68                      | 0.05                       | 1.79                | 1.35                      | 1.99                       |
| Phakellia excutia<sup>c</sup> | 0                         | 0.49                       | 1.63                | 1                        | 1.82                       |
| Trikenion flabelliforme       | 0.55                      | 0.11                       | 1.6                 | 0.93                      | 1.78                       |
| Cinachyra sp. BR21<sup>b</sup> | 0                         | 0.52                       | 1.54                | 1                        | 1.72                       |
| Cacospongia sp. SS4           | 0.61                      | 0.12                       | 1.52                | 0.99                      | 1.7                          |
| Sarcoragi sp. PB1<sup>b</sup> | 0.47                      | 0                          | 1.5                 | 0.99                      | 1.67                       |
| Ircinia sp. 1                 | 0.32                      | 0.46                       | 1.49                | 1.12                      | 1.66                       |
| Axinella arunsi Type 1        | 0.5                       | 0.06                       | 1.48                | 0.93                      | 1.65                       |
| Cymbastela stipitata          | 0.5                       | 0.26                       | 1.42                | 1.08                      | 1.59                       |
| Aplysina sp. 1                | 0.31                      | 0.38                       | 1.41                | 0.92                      | 1.58                       |
| Sarcoragi sp. PB2<sup>b</sup> | 0                         | 0.5                        | 1.31                | 0.64                      | 1.46                       |
| Oceanapia sp. SS13            | 0.48                      | 0.06                       | 1.23                | 0.82                      | 1.37                       |
| Axinella sp. 4                | 0.08                      | 0.36                       | 1.15                | 0.72                      | 1.29                       |
| Hyattella intestinalis cf.    | 0.09                      | 0.31                       | 1.11                | 0.66                      | 1.24                       |
| Cinachyraella australiensis cf.| 0.22                      | 0.33                       | 1.08                | 0.93                      | 1.21                       |
| Porphyria sp. PB1             | 0.05                      | 0.38                       | 1                   | 0.69                      | 1.12                       |
| Ianthea basis                 | 0.39                      | 0.06                       | 1                   | 0.89                      | 1.12                       |
| Amphimedon pararivéris cf.<sup>a</sup> | 0.4                 | 0                          | 0.99                | 0.66                      | 1.11                       |
| Halicosa sp. 10               | 0.19                      | 0.22                       | 0.99                | 0.72                      | 1.12                       |
| Mycale sp. 5                  | 0.08                      | 0.31                       | 0.99                | 0.5                      | 1.1                        |
| Clathria major<sup>c</sup>    | 0.37                      | 0                          | 0.94                | 0.84                      | 1.05                       |
| Ectyoplasia tuberculosis      | 0.21                      | 0.21                       | 0.92                | 0.78                      | 1.03                       |
| Amphimonas sulphurea          | 0.28                      | 0.16                       | 0.92                | 0.85                      | 1.03                       |
| Clathria lendenfeldi<sup>c</sup> | 0                          | 0.3                        | 0.91                | 0.53                      | 1.02                       |
| Spheciospongia sp. PB1<sup>b</sup> | 0.4                    | 0                          | 0.9                 | 0.81                      | 1.01                       |
| Rencoochela sp. 2             | 0.31                      | 0.06                       | 0.89                | 0.88                      | 1                          |
| Stylosa flabelliformis        | 0.2                       | 0.13                       | 0.87                | 0.65                      | 0.97                       |
| Clione sp. PB1                | 0.26                      | 0.1                        | 0.84                | 0.9                      | 0.94                       |
| Ciocalypta tyleri<sup>a</sup> | 0.19                      | 0                          | 0.82                | 0.52                      | 0.92                       |
| Acanthella cavernosa cf.<sup>c</sup> | 0                     | 0.28                       | 0.81                | 0.69                      | 0.91                       |
| Rhobadastrella globostellata cf.<sup>a</sup> | 0              | 0.3                        | 0.77                | 0.7                      | 0.86                       |
| Sphiceriopia testudinaria     | 0.29                      | 0.06                       | 0.75                | 0.75                      | 0.84                       |
| Iothoclea sp. KMB1            | 0.17                      | 0.06                       | 0.73                | 0.59                      | 0.82                       |
| Oceanapia sp. 7               | 0                         | 0.27                       | 0.73                | 0.45                      | 0.81                       |
| Sphiceriopia sp. K1 cf.       | 0.2                       | 0                          | 0.72                | 0.58                      | 0.8                        |
| Clathria abietina             | 0.24                      | 0.05                       | 0.71                | 0.74                      | 0.79                       |
| Flabelliss sp. PB1<sup>c</sup> | 0.11                      | 0                          | 0.7                 | 0.26                      | 0.78                       |
| Hyrtios sp. PBE<sup>c</sup>   | 0                         | 0.22                       | 0.68                | 0.56                      | 0.77                       |
| Dysisidae sp. 3               | 0.1                       | 0.14                       | 0.68                | 0.5                       | 0.76                       |

<sup>a</sup> Represents species which were found only pre-dredging.
<sup>b</sup> Represents species which were found only post-dredging.

Table 8
Summary table of total number of species (S), Simpson diversity (1 − λ'), Pielou's evenness (J'), average quantitative taxonomic diversity (Δ) and average quantitative taxonomic distinctness (Δ*). Note that Dive 5 was omitted from the analyses as it was considered to be a sponge depauperate habitat.

| Dredge period | S  | 1 − λ' | J' | Δ  | Δ*  |
|---------------|----|--------|----|----|-----|
| Pre           | 101| 0.985  | 0.9136| 74.75| 75.88 |
| Post          | 90 | 0.982  | 0.9071| 75.91| 77.3 |

4.1. Natural and dredging-related turbidity and sedimentation patterns

The nearshore environment of the study area is naturally turbid and located close (8 km) to the Ashburton River, which flooded on several occasions during the dredging phase (see Fig. 1). Before dredging, light penetration was low nearshore, with DLIs ranging from 1.3–2.0 mol photons m⁻² d⁻¹ (P₉₀₅, 1–30 d running mean interval) reaching the benthos at 9 m depth (see ROLLER). There is a natural inshore to offshore turbidity gradient which is also likely to be a consequence of wind and wave resuspension of the gradually shallowing seabed closer to the shore (see Supplementary Fig. 5 for Landsat images of the study area through time). Dredging was associated with development of a coastal LNG facility and shore side berth pockets, jetty, material off-loading facilities, a turning basin, and also a long (~16 km) navigation channel. The dredging had clear and pronounced effects on water column turbidity as has been reported in several other dredging...
Based on an 80th percentile of the 1 d running means, the ratio of dredging/baseline turbidity ranged between 0.9 and 2.5. This ratio is lower than 3 other large scale capital dredging projects in Western Australia, where values ranged between 0.5–6.8 (Barrow Island project), 1.3–4.8 (Burrup Peninsula project), and 1.5–18.6 (Cape Lambert project, see Fisher et al., 2015).

The difference in dredging/baseline NTUs between projects is likely to be due to a number of factors including the proximity of the monitoring sites from dredging, which was in turn related to the location of areas of interest, such as individual shoals or reefs of preservation value. In addition, intrinsic differences between projects, for example concentrated point-source dredging of material offloading and turning basins characteristic of the Barrow Island, Burrup Peninsula and Cape Lambert projects, compared to the dispersed dredging operation along a 16 km navigation channel at Wheatstone (making up ~70% of total dredging), may have contributed to the comparatively lower NTU during the dredging phase at Onslow (Jones et al., 2015a; Fisher et al., 2015). Lastly, control measures were adopted by the dredging proponent to reduce water quality impacts for the Wheatstone project, which include limiting overflows from trailing suction hopper dredges, and the use of backhoe dredges in more environmentally sensitive areas (Chevron, 2009, 2012).

Fig. 7. Summary of mean Chl-a content (μg g⁻¹ ww sponge) of all sponge species (n = 149) collected during the pre- and post-dredging sampling events. Blue vertical bars represent values derived from samples collected pre-dredging and red vertical bars represent values derived from samples collected post-dredging. Dashed grey and black horizontal lines represent Chl-a levels of 32.9 and 63.5 μg g⁻¹ ww sponge corresponding to the low and high limits of photosynthetic capabilities of sponges respectively, derived from Wilkinson (1983). Dashed red horizontal line represents Chl-a level of 2.6 μg g⁻¹ ww sponge derived from a known heterotrophic sponge *Ianthella basta* in this study, and values falling below this represent no photosynthetic activity. Horizontal coloured bars underneath the graph represent periods when sponge species were found; blue (pre-dredging only), yellow (pre- and post-dredging) and red (post-dredging only). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Australia, where values ranged between 0.5–6.8 (Barrow Island project), 1.3–4.8 (Burrup Peninsula project), and 1.5–18.6 (Cape Lambert project, see Fisher et al., 2015).

The difference in dredging/baseline NTUs between projects is likely to be due to a number of factors including the proximity of the monitoring sites from dredging, which was in turn related to the location of areas of interest, such as individual shoals or reefs of preservation value. In addition, intrinsic differences between projects, for example concentrated point-source dredging of material offloading and turning basins characteristic of the Barrow Island, Burrup Peninsula and Cape Lambert projects, compared to the dispersed dredging operation along a 16 km navigation channel at Wheatstone (making up ~70% of total dredging), may have contributed to the comparatively lower NTU during the dredging phase at Onslow (Jones et al., 2015a; Fisher et al., 2015).

Lastly, control measures were adopted by the dredging proponent to reduce water quality impacts for the Wheatstone project, which include limiting overflow from trailing suction hopper dredges, and the use of backhoe dredges in more environmentally sensitive areas (Chevron, 2009, 2012). Surveys of the particle size distribution (PSD) of the seabed 1 month after the dredging showed there had been a marked increase in the fines content (< 60 μm) of sediments compared to pre-dredging surveys. The
post-dredging survey also identified a clear gradient of decreasing silt content with increasing distance, up to the furthest site 1.5 km away from the areas of dredging. The gradient is mostly likely to have been caused by the release of sediments (spillage) into the water column, advection of the plumes away from the site of dredging and subsequent settlement of the sediment in a deposition zone. A follow-up survey 7 months after the dredging indicated slightly coarser PSDs, and a return towards pre-dredging levels; nevertheless, the gradient and sediment deposition zone still remained. The 7 month post-dredging survey was conducted after a category 3 cyclone (Tropical Cyclone Olwyn) passed through the study area, producing wind gusts of up to 140 km h$^{-1}$, very heavy seas and profoundly disturbing the local environment. The cyclone is likely to have contributed to the change in PSDs by preferentially resuspending and displacing the fines. Increases in the silt content of sediments after dredging projects have previously been reported by Jones et al. (2016). Increased siltification of the environment appears to be a long term legacy of large scale dredging projects, and how long it takes to return to pre-dredging levels are unknown.

4.2. Patterns of benthic communities

There is some evidence that dredging influenced benthic communities; however, the effect was relatively weak, with taxa specific responses including both decreases and increases in abundances. It is important to note that the post-dredging surveys were conducted in July 2015 when the dredging was completed (February 2015), but after Cyclone Olwyn passed through the area (March 2015). As with the sediment particle size analysis described above, the swells and sea associated with the cyclone could have affected the abundance and benthic community composition in addition to the dredging campaign.

Macroalgae showed the most prominent reduction in abundance across all channel zones, including up to 96% at Inner channel zone. This may have been caused by light attenuation associated with the elevated turbidity experienced through periods of dredging and cyclone, as reported for other phototrophic taxa such as seagrass (Collier et al., 2012). Sedimentation has been shown to reduce the abundance of some macroalgal species by limiting growth and survival of newly settled life history stages (e.g. spores or gametes, Eriksson and Johansson, 2005), and may also be a plausible cause-effect pathway in the observed reduction in macroalgae abundance. However, since macroalgae can be very seasonal in distribution (Vuki and Price, 1994; Plouguerné et al., 2006; Abdul Wahab et al., 2014a), an effect of sampling at different times of year cannot be discounted.

Individual assessments of filter feeding taxa showed a significant reduction of the hydrozoans, with up to 97% reduction in abundance at the Inner and Outer channel zones. Detrimental response to increased sedimentation has been reported in other suspension feeders, such as barnacles, where clogging of the feeding apparatus (i.e. cirri) led to mortality within 1 h of exposure (Fabricius and Wolanski, 2000), thus similar sensitivities to increased sediment load may explain the observed decline in hydrozoans. While colonial ascidian abundance was generally stable within Inner and Mid zones, a 20-fold reduction in abundance was seen at the Outer channel zone which may have been due to the increase in turbidity. While sediment clearing behaviours have been described for ascidians, the reduction in abundance post-dredging could have been due to burial and clogging of siphons and branchial walls in sediment sensitive species (Armsworthy et al., 2001). Ascidians have been shown to be affected by increased sediment loading in the environment; however, whether the response is negative, positive or neutral is highly species-specific (Naranjo et al., 1996), and ascidians were not taxonomically identified in this study.

Hard corals showed a slight increase in abundance after the dredging which is notable given the potential sensitivity of the early life history stages of corals to sediments and dredging (Rogers, 1990, Fabricius, 2005, Jones et al., 2015b). Corals in the area have been impacted from several bleaching events and cyclones in recent years. In particular, in February–March 2011 (before dredging) the study area experienced a period of unprecedented sea surface temperatures that caused widespread coral bleaching and fish kills along the west coast of Australia (Thomson et al., 2011; Moore et al., 2012; Smale and Wernberg, 2012; Depczynski et al., 2013; Feng et al., 2013; Zinke et al., 2014; Ridgway et al., 2016). The area also experienced a period of intense storms and cyclonic activity with 2 cyclones (Bianca and Carlos) passing close by the study site in January and February 2011 respectively (Moore et al., 2012; Ridgway et al., 2016). High declines in coral cover occurred during this period, falling from 29–68% (mean 45%) to < 10%, at > 90% of those reefs monitored before 2011. In addition, in early 2013, immediately before dredging started, the study area experienced another period of elevated seawater temperatures which resulted in > 50% of the remaining hard corals bleached (Lafratta et al., 2016). Despite these disturbances (and cyclone Olwyn in March 2015), there was a small increase in the number of corals per m² over the dredging period, which may indicate recovery of hard coral populations following these high mortality events.

4.3. Patterns of sponge functional morphology

The sensitivity of sponges to sedimentation is known to vary between taxa with different morphologies (de Voogd and Cleary, 2007). However, the existing patterns in sponge functional morphology remained relatively stable through periods of elevated sediment levels, which indicate an established community adapted to living in environments exposed to high sediment load. The high abundance of encrusting morphotypes pre-dredging was unexpected considering their low morphological profile which could correspond to higher susceptibility to detrimental sedimentation and smothering in a naturally turbid habitat influenced by river discharge plumes. However, sediment responses and active sediment clearing mechanisms exist in sponges which include alteration and cessation of pumping (Tompkins-MacDonald and Leys, 2008), flow reversal (Simpson, 1984) and mucus production (Bannister et al., 2012), which could have contributed to individual survival during intermittent periods of high turbidity and sedimentation.

The persistence of encrusting and endolithic sponge forms to sedimentation has been reported in other tropical areas (Carballo, 2006). The high reduction in abundance in encrusting morphotypes post-dredging, suggest that chronic exposure to sedimentation over the ~2 years of the dredging program may have incurred a physiological energetic deficit through constant active sediment clearing, thus contributing to mortality of individuals. Interestingly, there were morphologies which appeared to have benefited from the increased levels of turbidity and sedimentation, with erect branching and massive cryptic morphotypes showing a 4-fold increase in numbers post-dredging. An erect and branching body plan would mean the chances of smothering would be low, as most of the sponge body is elevated off the benthos with minimal horizontal surface area available for smothering or burial. In massive cryptic morphotypes, most of the sponge is buried in sediment, with fistules (protruding finger-like structures) crucial in facilitating pumping above the sediment during burial (see Bell et al., 2015 and Schönberg, 2015 for reviews on sponge adaptations to sediment). These morphologies represent passive adaptations to living in areas subject to high sediment, and have been reported previously in fan (erect laminar) and fistulose (massive cryptic) sponges in the Indo-Pacific (de Voogd and Cleary, 2007).

Cup morphotypes were underrepresented in the study area, which could be attributed to susceptibility of this morphology to accumulating sediment, and exposure to associated physiological stress due to sedimentation (Pineda et al., 2016a). Photosynthetic sponges (e.g. Carteriospongia and Phillospongia spp.), which can form up to 80% of sponges on clear water reefs of the Great Barrier Reef (GBR), are typically cup-shaped or foliose, a morphology which facilitates light capture for...
photosynthesis. These species can be depth limited if turbidity is high (Wilkinson, 1983, 1988; Wilkinson and Evans, 1989; Webster et al., 2012; Abdul Wahab et al., 2014b). Both *Carteriospongia* and *Phylllospongia* spp. have been reported from other localities in the Pilbara Nearshore bioregion and neighbouring Pilbara Offshore bioregion, but was absent at the study site (Abdel Wahab et al., 2014c; Fromont et al., 2016).

Pre-dredging PD 30 DLI at the 30 d running mean interval around sponge survey sites in this study ranged between 1.9 and 5.0 (ASHNE, PAROO, ENDCH, SALD and GORGSW). In a 28 d experiment investigating the effects of light on 3 phototropic and 2 heterotrophic sponges, Pineda et al. (2016b) reported no detrimental effects of light attenuation, up to 0 DLI, on the heterotrophic *Styloides flabelformis* and *Ianthella basta*. Both of these species were found at pre- and post-dredging surveys in this study. Sister species of two of the phototropic sponges *Cliona orientalis* (C. sp. PB1 in this study) and *Cymbastela coralliophila* (*C. stipitata* in this study) were also found during the pre- and post-dredging surveys, corroborating resilience of these species under low light conditions (Pineda et al., 2016b). Interestingly, the highly phototropic *Carteriospongia foliascens* exposed to low experimental DLI levels (< 0.8), showed bleaching and mortality, with no mortality and positive growth recorded at DLI ≥ 3.2 under the 28 d experimental conditions (Pineda et al., 2016b). Historical exposure (> 2 y) to natural DLI levels between 1.9 and 5.0, interspersed with extended periods of high turbidity (corresponding to PD DLI at 30 d running mean interval ranging between 0.2 and 3.1), may incur mortalities to adults and new recruits of *C. foliascens* arriving to the area. This might explain the absence of *Carteriospongia* spp., and suggests the turbid environment of Onslow to be a poor habitat for highly phototropic sponges. Notably, natural environmental filtering may have selected for morphologies and traits tolerant to turbidity and sedimentation stress and may explain the stability of the functional morphology assemblage post-dredging. Assemblages of sponges with cup morphologies, either heterotrophic or phototropic, may be good indicators for inferring historic turbidity and sedimentation regime of a habitat, and could be useful in environmental impact assessment processes for managing dredging related sediment pressures in clear water habitats.

4.4. Patterns of sponge species composition

The high recovery of new sponge records (Linnaean species and OTUs; up to > 50% of all sponges collected), for Onslow and the IMCRA Pilbara Nearshore bioregion, over the entire study, clearly indicates an effect of undersampling in the area. Therefore, comparisons of species compositions between sampling periods are most likely unreliable in detecting responses to sediment pressures. Nevertheless, the reporting of 150 sponge species is important as it shows that despite natural and dredging related sediment pressures, Onslow supports relatively high sponge diversity which potentially constitutes species adapted to living in highly turbid habitats.

While species data may not be reliable in inferring community level changes, species-specific assessments may provide insights into taxa which are potentially influenced by the increased sediment loading. For example, *Raspailia keriontria* which occurred at the highest abundance under elevated sediment levels while not detected at lower sediment levels during baseline surveys suggests this species may possess sediment tolerance mechanisms which allow it to thrive under increased sediment loading. The opposite was observed for some species such as *Axinella aruensis* Type II and *Psammocinia cf. bulbosa*, where they occurred in high abundances pre-dredging but were not detected post-dredging, which suggests higher sensitivities of these species to sediment. Notably, increased sediment perturbations have been shown to alter species composition for sponges and other invertebrate taxa in other tropical areas, whereby certain species thrive and dominate the benthos under conditions where other species seem to be negatively affected, thus highlighting species specific sensitivity of sponges to sediment related stress (Norström et al., 2009; Knapp et al., 2013).

4.5. Community response around the site of highest water quality impact

The most seaward extent of the navigation channel is an area of particular interest as the communities there are least likely to be adapted to naturally high turbidity levels associated with the natural inshore to offshore turbidity gradient. The End of Channel (ENDCH) water quality monitoring site, which was located ~700 m from the dredge channel, recorded the highest increase of turbidity above baseline levels, and one of the highest absolute turbidity levels during dredging. It also experienced both short term acute NTU elevations when dredging occurred nearby, as well as low level chronic elevations associated to westerly drift of sediment from the dredge material placement sites. The increase in turbidity also resulted in reduced DLI to the benthos at 0.4× of pre-dredging levels. Notably, areas surrounding ENDCH was designated as a “zone of high impact”, indicating all loss of benthic communities was allowed under the environmental approval permits and where limited management of water quality was needed (Chevron, 2012).

Detailed analyses of benthic communities and sponge functional morphologies around the ENDCH monitoring site (300 m to 1.5 km away), showed similar changes as described for the coarser scale analyses for the Outer channel zone. These include an increase in the number of sponges, gorgonians and hard corals, a reduction in the number of hydrozoans and a marked reduction in colonial ascidians. This study was unable to determine if the reduction in numbers of taxa were unequivocally attributed to dredging-related turbidity, as opposed to possible additive effects of cyclone Olwyn. Similarly, it is difficult to determine whether the increases, which potentially are associated to recovery trajectory from past thermal stress events, would have been higher if dredging had not occurred. Nevertheless, despite an approximate doubling of turbidity over the ~2 y of dredging, including episodic intermittent peaks in turbidity when dredging was occurring nearby, there was a relative stability in sponge, hard coral and gorgonian numbers, and sponge community functional morphologies.

5. Conclusion

Water quality was continually monitored during the dredging program and used in an adaptive management framework to limit the intensity and duration of disturbances. Nevertheless, there were still pronounced acute and chronic changes in water quality over large areas and in some locations (where there was no requirement to manage water quality) turbidity levels approximately doubled and light levels halved over the dredging. Despite these changes, and also the influence of a marine heatwave, tropical cyclones and several flooding events, there were no marked effects on sponge abundance, morphology and mode of nutrition. This apparent stability shows a degree of resilience of sponge communities to water quality disturbances during a well-managed dredging program, and may also indicate an established sponge community adapted to living in environments characterized by high sediment loads and regular cyclone exposure. This highlights the importance of considering historic disturbance regimes of communities when assessing impacts from dredging.

Author contributions

RJ and MAAW conceived the study. MAAW, JF and OG performed field surveys, sample collections and data collation. MAAW managed chlorophyll extractions and analyses. JF and OG conducted detailed taxonomic identifications for sponges. RF managed QAQC of turbidity data and provided turbidity plots and intellectual inputs on statistical analyses. MAAW analysed the data in detail and produced figures assisted by JF, OG, RF and RJ, and MAAW, JF, RF and RJ wrote the paper.
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### Supplementary Table 1: Summary table of towed video surveys, including GPS coordinates of tow transect midpoints, associated channel zones, mean depth of towed survey, number of images taken, and mean depth and SE for channel zones

| Tow number | Channel zone | Midpoint latitude | Midpoint longitude | Pre-dredging (2013) | Post-dredging (2015) |
|------------|--------------|-------------------|--------------------|----------------------|-----------------------|
|            |              |                   |                    | Tow mean depth (m)   | n images  | Zone mean depth (m) | Zone SE depth |
|            |              |                   |                    |                      |          |                      |              |
| 6          | Outer        | -21.534464        | 115.027324         | 13.6                 | 48        | 13.6                 | 0.2          |
| 9          | Outer        | -21.528056        | 115.026961         | 13.6                 | 52        | 13.4                 | 0.2          |
| 10         | Outer        | -21.542228        | 115.050067         | 12.7                 | 56        | 12.1                 | 0.2          |
| 12         | Outer        | -21.536369        | 115.050784         | 13.1                 | 67        | 12.8                 | 0.2          |
| 13         | Outer        | -21.539182        | 115.046354         | 13.6                 | 62        | 12.0                 | 0.2          |
| 14         | Outer        | -21.537495        | 115.043176         | 13.6                 | 58        | 12.8                 | 0.2          |
| 15         | Outer        | -21.537687        | 115.038908         | 13.4                 | 67        | 11.9                 | 0.2          |
| 18         | Outer        | -21.536697        | 115.023587         | 13.4                 | 67        | 12.0                 | 0.2          |
| 22         | Outer        | -21.51949         | 115.037396         | 14.4                 | 45        | 14.7                 | 0.2          |
| 27         | Outer        | -21.529531        | 115.051215         | 14.2                 | 56        | 14.0                 | 0.2          |
| 28         | Outer        | -21.521129        | 115.038539         | 14.4                 | 55        | 14.2                 | 0.2          |
| 29         | Mid          | -21.573042        | 114.997034         | 10.2                 | 57        | 9.6                  | 0.2          |
| 30         | Mid          | -21.576655        | 115.002419         | 10.6                 | 60        | 10.4                 | 0.2          |
| 31         | Mid          | -21.584329        | 115.000595         | 10.6                 | 68        | 10.8                 | 0.2          |
| 32         | Mid          | -21.58296         | 114.994056         | 10.3                 | 45        | 10.4                 | 0.1          |
| 33         | Mid          | -21.585126        | 115.008265         | 10.7                 | 55        | 10.8                 | 0.2          |
| 34         | Mid          | -21.582983        | 115.011997         | 10.2                 | 49        | 10.0                 | 0.2          |
| 35         | Mid          | -21.58421         | 115.01148          | 10.2                 | 50        | 10.3                 | 0.2          |
| 56         | Inner        | -21.632715        | 115.006343         | 7.9                  | 66        | 8.6                  | 0.2          |
| 57         | Inner        | -21.62036         | 115.010957         | 8.4                  | 98        | 9.1                  | 0.2          |
| 58         | Inner        | -21.621291        | 115.013889         | 8.0                  | 68        | 9.2                  | 0.2          |
| 60         | Inner        | -21.616979        | 114.981515         | 10.4                 | 51        | 10.4                 | 0.2          |
| 63         | Inner        | -21.619473        | 114.990505         | 9.6                  | 66        | 9.3                  | 0.3          |
| 64         | Inner        | -21.627602        | 115.004887         | 9.3                  | 54        | 9.0                  | 0.3          |
| 65         | Inner        | -21.627903        | 115.011588         | 9.5                  | 70        | 8.7                  | 0.3          |
| 70         | Inner        | -21.620673        | 114.987306         | 9.6                  | 94        | 9.3                  | 0.3          |
| 72         | Inner        | -21.618331        | 114.984855         | 10.7                 | 109       | 9.8                  | 0.3          |
**Supplementary Table 2:** Summary information of tropical cyclones which passed in the proximity of the Onslow study area within the study period. The first five cyclones (Iggy, Lua, Mitchell, Rusty and Christine) passed within two years prior to commencement of dredging and two years while dredging was conducted, while Cyclone Olwyn passed three months after dredging was completed. NTU on days for when cyclones passed through the study area were omitted from the dataset (Days Removed) to remove resulting NTU spikes which can influence assessment of long-term, chronic turbidity effects from dredging. NA – not applicable.

| # | Tropical cyclone | Cyclone max category | Year   | Start       | End         | Days removed | Comments                        |
|---|------------------|----------------------|--------|-------------|-------------|--------------|---------------------------------|
| 1 | Iggy             | 2                    | 2012   | 22/01/2012  | 17/02/2012  | 26           | nearest gales 150+ km away      |
| 2 | Lua              | 4                    | 2012   | 9/03/2012   | 1/04/2012   | 23           | nearest gales 17 km away        |
| 3 | Mitchell         | 1                    | 2013   | 27/12/2012  | 15/01/2013  | 19           | nearest gales 40 km away        |
| 4 | Rusty            | 4                    | 2013   | 22/02/2013  | 14/03/2013  | 20           | nearest gales 140 km away, large system |
| 5 | Christine        | 4                    | 2014   | 25/12/2013  | 15/01/2014  | 21           | nearest gales <10 km away       |
| 6 | Olwyn            | 3                    | 2015   | 08/03/2015  | 14/03/2015  | NA           | nearest gales 0 km away         |
### Supplementary Table 3: Summary of daily light integral (DLI) percentile values (\(P_{10}, P_{20}, P_{30},\) and \(P_{50}\)) for running mean intervals of 1, 7, 14 and 30 d for the 16 water monitoring sites for the Wheatstone dredging project. The ratio of dredging/ baseline NTUs at the 20th percentile (\(P_{20}\)) is provided in the last column. Cell colours represent relative strength of NTU and dredging/ baseline ratio values, with dark green representing high values and dark red representing low values.

| Site  | Pre-dredging | Before 1 d | Before 7 d | Before 14 d | Before 30 d | During 1 d | During 7 d | During 14 d | During 30 d | Ratio change \(P_{20}\) |
|-------|---------------|------------|------------|-------------|-------------|------------|------------|-------------|-------------|------------------|
| BESS  | 1             | 8.7 6.9 4.4 0.4 | 8.8 7.1 4.5 2.5 | 8.7 7.2 4.3 3.2 | 9.3 8.4 5.0 3.7 | 7.6 5.5 4.2 1.1 | 7.3 5.9 5.2 3.1 | 7.2 6.1 5.3 3.9 | 7.3 6.3 5.3 4.1 | 0.9 0.8 0.8 0.8 |
| THISE | 1             | 4.2 3.1 2.2 0.2 | 4.6 3.4 2.6 0.3 | 4.8 3.9 3.2 0.4 | 5.0 4.4 2.1 1.6 | 2.1 1.3 0.9 0.2 | 2.3 1.8 1.3 0.4 | 2.4 2.0 1.8 0.4 | 2.4 2.1 2.0 1.3 | 0.5 0.5 0.5 0.5 |
| ASHIE | 2             | 4.1 3.1 2.2 0.0 | 4.5 3.2 2.7 0.8 | 4.5 3.3 2.8 1.6 | 4.6 3.5 3.3 3.1 | 1.7 1.0 0.6 0.0 | 1.8 1.5 1.1 0.2 | 1.9 1.7 1.5 0.5 | 2.0 1.9 1.8 1.5 | 0.4 0.4 0.4 0.4 |
| AYLIE | 2             | 6.9 5.3 3.9 0.6 | 7.2 5.8 5.1 3.1 | 6.9 6.4 5.7 4.6 | 7.0 6.3 6.2 5.8 | 4.8 3.3 2.5 0.3 | 4.9 3.8 3.3 1.2 | 5.1 4.1 3.5 2.8 | 5.1 4.3 4.0 3.3 | 0.7 0.7 0.7 0.7 |
| THIE  | 2             | 3.7 2.8 2.3 0.4 | 4.2 3.3 2.7 2.0 | 4.6 3.5 3.2 2.3 | 4.7 3.8 3.7 3.3 | 2.4 1.7 1.0 0.3 | 2.7 2.1 1.6 0.3 | 2.9 2.1 1.7 0.9 | 2.9 2.2 1.6 1.3 | 0.6 0.6 0.6 0.6 |
| WEEKS | 2             | 1.4 0.6 0.2 0.0 | 1.7 0.9 0.4 0.1 | 2.0 1.4 0.5 0.1 | 2.4 2.0 1.5 0.2 | 0.5 0.3 0.2 0.0 | 0.6 0.5 0.4 0.1 | 0.6 0.6 0.4 0.2 | 0.7 0.6 0.5 0.3 | 0.4 0.4 0.3 0.3 |
| PAROD | 2             | 4.0 2.7 1.9 0.7 | 4.3 3.1 2.4 1.6 | 4.5 3.2 2.6 2.1 | 5.0 3.9 3.5 2.6 | 1.4 0.8 0.4 0.0 | 1.7 1.0 0.8 0.3 | 1.8 1.3 1.0 0.7 | 1.8 1.5 1.3 0.9 | 0.4 0.4 0.4 0.4 |
| ENDCH | 2             | 1.4 0.7 0.3 0.0 | 1.6 0.9 0.4 0.1 | 1.9 1.1 0.6 0.1 | 1.9 1.6 1.3 0.2 | 0.5 0.2 0.1 0.0 | 0.6 0.4 0.3 0.1 | 0.7 0.5 0.3 0.2 | 0.7 0.5 0.4 0.2 | 0.3 0.4 0.4 0.4 |
| SALAD | 2             | 2.7 1.8 1.2 0.1 | 3.1 2.2 1.8 0.7 | 3.2 2.6 2.2 1.3 | 3.6 2.7 2.7 2.4 | 0.9 0.5 0.3 0.0 | 1.2 0.7 0.6 0.3 | 1.2 0.9 0.7 0.4 | 1.2 1.1 0.9 0.7 | 0.4 0.4 0.4 0.4 |
| HERALD| 3             | 4.9 3.4 2.5 0.3 | 5.3 3.9 2.9 1.4 | 5.5 4.1 3.5 2.2 | 6.0 4.6 3.9 2.9 | 1.6 1.2 0.8 0.0 | 1.7 1.3 1.1 0.4 | 1.7 1.4 1.2 0.9 | 1.9 1.5 1.3 1.2 | 0.3 0.3 0.3 0.3 |
| TWIN  | 3             | 4.1 3.0 2.2 0.4 | 4.6 3.3 2.7 1.4 | 5.0 3.9 3.0 2.2 | 4.6 4.1 3.7 3.3 | 3.1 1.8 1.2 0.0 | 3.5 2.6 1.8 0.3 | 3.7 3.0 1.4 1.9 | 4.1 3.4 2.8 1.9 | 0.7 0.8 0.8 0.9 |
| GORGSW| 3             | 2.6 1.9 1.2 0.2 | 2.9 2.2 1.7 0.8 | 3.2 2.4 1.9 1.2 | 3.1 2.7 2.6 2.4 | 1.1 0.7 0.4 0.0 | 1.3 1.0 0.8 0.2 | 1.4 1.2 1.1 0.8 | 1.4 1.3 1.2 1.0 | 0.4 0.5 0.4 0.5 |
| WEST  | 3             | 2.8 1.8 1.3 0.3 | 3.2 2.6 2.2 0.6 | 3.5 3.2 3.0 1.9 | 3.6 3.2 2.9 2.6 | 1.2 0.7 0.5 0.0 | 1.7 0.9 0.7 0.3 | 1.9 1.1 0.7 0.6 | 2.1 1.7 0.8 0.6 | 0.4 0.5 0.6 0.6 |
| DIRNE | 3             | 4.3 2.9 2.0 0.1 | 4.3 3.4 2.5 1.2 | 4.7 3.7 3.0 1.6 | 5.1 4.0 3.6 2.7 | 3.1 2.1 1.1 0.0 | 3.4 2.7 1.9 0.2 | 3.4 2.8 2.4 1.2 | 3.5 3.0 2.6 2.1 | 0.8 0.8 0.7 0.7 |
| WARD  | 4             | 1.9 1.4 0.8 0.0 | 2.5 1.9 1.4 0.6 | 2.7 2.4 2.0 1.2 | 3.1 3.0 2.8 2.1 | 0.9 0.5 0.1 0.0 | 1.5 1.1 0.6 0.0 | 1.7 1.4 1.0 0.2 | 1.8 1.4 1.2 0.9 | 0.5 0.6 0.6 0.6 |
| ROLLER| 4             | 1.3 0.8 0.4 0.0 | 1.7 1.2 0.9 0.3 | 1.9 1.4 1.1 0.8 | 2.0 1.7 1.5 1.1 | 0.5 0.2 0.1 0.0 | 0.8 0.4 0.2 0.0 | 1.0 0.5 0.4 0.1 | 1.3 0.6 0.5 0.3 | 0.4 0.5 0.5 0.6 |
**Supplementary Table 4**: List of sponge Linnean species and operational taxonomic unit (OTUs) recorded from the Onslow study area pre- and post-dredging, and corresponding mean and SE densities (individuals 5m⁻²). Text in red identifies where taxa were absent pre- or post-dredging. Presence/absence (1/0) of species at six Interim Marine and Coastal Regionalisation for Australia (IMCRA) bioregions within the Pilbara area derived from the study of Fromont et al. (2016) are provided for comparison of species distributions. The "Record" column lists species and OTU records: 1) new to the Pilbara region (α); 2) new to the IMCRA Pilbara Nearshore bioregion (PIN, β) and 3) previously reported from the Pilbara Nearshore bioregion (γ). IMCRA bioregion abbreviations: NWS – North West Shelf, PIO – Pilbara Offshore, PIN – Pilbara Nearshore, NIN – Ningaloo, NWP – North West Province and CWT – Central West Transition.

| #  | Class       | Order            | Family       | Species                                      | NWS Record density | PIN Record density | NIN Record density | NWP Record density | CWT Record density |
|----|-------------|------------------|--------------|----------------------------------------------|---------------------|---------------------|---------------------|---------------------|---------------------|
| 1  | Calcarea    | Calcarea unknown | Calcarea unknown | Calcarea sp. PB2 | 0 | 0 | 0 | 0 | 0.023 |
| 2  | Calcarea    | Clathrinida      | Leucettida   | Leucetta sp. PB1 | 0 | 0 | 0 | 0 | 0.045 |
| 3  | Calcarea    | Clathrinida      | Leucettida   | Leucetta sp.white | 0 | 0 | 0 | 0.023 | 0 |
| 4  | Demospongia | Agelasida        | Agelasida    | Agelas cf. mauritiana | 0 | 0.068 | 0.049 | 0.045 |
| 5  | Demospongia | Agelasida        | Agelasida    | Amphinomia sulphera | 1 | 0.295 | 0.161 | 0.091 |
| 6  | Demospongia | Axinellida       | Axinellida   | Axinella aruensis Type I | 0 | 0.500 | 0.199 | 0.045 |
| 7  | Demospongia | Axinellida       | Axinellida   | Axinella cf. aruensis Type I | 0 | 0.091 | α |
| 8  | Demospongia | Axinellida       | Axinellida   | Axinella aruensis Type II | 0 | 0.101 | 0.315 | 0 |
| 9  | Demospongia | Axinellida       | Axinellida   | Axinella sp. 4 | 0 | 0.068 | 0.068 | 0.409 |
| 10 | Demospongia | Axinellida       | Heteroxyida  | Myrmekioderma granulatum | 0 | 0.068 | 0.049 | 0 |
| 11 | Demospongia | Axinellida       | Raspaliida   | Ceratopision axefor | 1 | 0.636 | 0.385 | 0.864 |
| 12 | Demospongia | Axinellida       | Raspaliida   | Ceratopision cf. montebelloensis | 0 | 0.114 | 0.091 | 0 |
| 13 | Demospongia | Axinellida       | Raspaliida   | Echinodictyum cancellatum | 1 | 0.114 | 0.052 | 0 |
| 14 | Demospongia | Axinellida       | Raspaliida   | Echinodictyum clathrioides | 1 | 0.159 | 0.070 | 0.023 |
| 15 | Demospongia | Axinellida       | Raspaliida   | Ectyoplasia frondosa | 0 | 0.068 | 0.035 | 0 |
| 16 | Demospongia | Axinellida       | Raspaliida   | Ectyoplasia tabula | 1 | 0.182 | 0.117 | 0.159 |
| 17 | Demospongia | Axinellida       | Raspaliida   | Ectyoplasia vannus | 0 | 0.023 | 0.091 | 0 |
| 18 | Demospongia | Axinellida       | Raspaliida   | Raspalia (Clathriodendron) kerloni | 0 | 1.477 | 0.416 | 0 |
| 19 | Demospongia | Axinellida       | Raspaliida   | Raspalia (Raspalia) vestigera | 0 | 0.068 | 0.035 | 0.023 |
| 20 | Demospongia | Axinellida       | Raspaliida   | Raspalia (Raspaxilla) sp. PB1 | 0 | 0.091 | 0.061 | α |
| 21 | Demospongia | Axinellida       | Raspaliida   | Reniochalin sp.2 | 0 | 0.227 | 0.098 | 0.045 |
| 22 | Demospongia | Axinellida       | Raspaliida   | Reniochalin stalgmitis | 1 | 0.977 | 0.357 | 0.091 |
| 23 | Demospongia | Axinellida       | Raspaliida   | Reniochalin cf. stalgmitis | 0 | 0.114 | 0.052 | 0 |
| 24 | Demospongia | Axinellida       | Raspaliida   | Solosalia digitate | 0 | 0.068 | 0.049 | 0.045 |
| 25 | Demospongia | Axinellida       | Raspaliida   | Thrinacophora cervicornis | 1 | 0.260 | 0.068 | 0.049 |
| 26 | Demospongia | Axinellida       | Raspaliida   | Trikentriani fiabelliforme | 1 | 0.591 | 0.260 | 0.068 |
| 27 | Demospongia | Axinellida       | Stelligerida  | Higginsia scabra | 1 | 0.205 | 0.146 | 0 |
| 28 | Demospongia | Axinellida       | Stelligerida  | Higginsia cf. scabra | 0 | 0.045 | 0 |
| 29 | Demospongia | Biemmida         | Biemmida     | Biemma cf. sp. 2 | 0 | 0.068 | 0.068 | 0 |
| 30 | Demospongia | Bubarida         | Bubarida     | Pararhaphoxy sp. PB1 | 0 | 0.045 | 0.030 | 0 | α |
| #  | Class        | Order       | Family      | Species                        | NWS | PIQ | PIN | NN | NWP | CVMT | Mean density | SE density | Mean density | SE density | Record |
|----|--------------|-------------|-------------|--------------------------------|-----|-----|-----|----|-----|------|---------------|------------|--------------|------------|--------|
| 31 | Demospongiae | Bubarida    | Dictyonellidae | Acanthella cf. cavernosa | 0   | 1   | 0   | 1  | 0   | 0    | 0.00          | 0.00       | 0.227        | 0.104      | β      |
| 32 | Demospongiae | Bubarida    | Dictyonellidae | Acanthella pulcherrima      | 0   | 0   | 0   | 1  | 1   | 0    | 0.00          | 0.00       | 0.114        | 0.091      | β      |
| 33 | Demospongiae | Bubarida    | Dictyonellidae | Acanthella cf. pulcherrima  | 0   | 1   | 1   | 0  | 0   | 0    | 0.00          | 0.00       | 0.114        | 0.091      | γ      |
| 34 | Demospongiae | Bubarida    | Dictyonellidae | Cymbastela stipitata       | 0   | 0   | 1   | 0  | 0   | 0    | 0.409         | 0.123      | 0.205        | 0.116      | γ      |
| 35 | Demospongiae | Bubarida    | Dictyonellidae | Phaketta euctimena        | 1   | 0   | 1   | 0  | 0   | 0    | 0.00          | 0.00       | 0.409        | 0.144      | γ      |
| 36 | Demospongiae | Chondrillarda | Chondrillardae | Chondrilla australiensis  | 0   | 0   | 1   | 1  | 0   | 0    | 0.00          | 0.00       | 0.068        | 0.049      | γ      |
| 37 | Demospongiae | Clionaidae  | Clionaidae | Cliona sp. PB1            | 0   | 0   | 0   | 0  | 0   | 0    | 0.159         | 0.061      | 0.114        | 0.114      | α      |
| 38 | Demospongiae | Clionaidae  | Clionaidae | Sphaciospongia cf. papillosa | 0   | 0   | 0   | 0  | 0   | 0    | 0.023         | 0.023      | 0.00         | 0.00 α     | α      |
| 39 | Demospongiae | Clionaidae  | Clionaidae | Sphaciospongia cf. sp. K1  | 0   | 0   | 0   | 0  | 0   | 0    | 0.114         | 0.052      | 0.00         | 0.00 α     | α      |
| 40 | Demospongiae | Clionaidae  | Clionaidae | Sphaciospongia sp. PB1     | 0   | 0   | 1   | 0  | 0   | 0    | 0.386         | 0.180      | 0.00         | 0.00 γ     | γ      |
| 41 | Demospongiae | Clionaidae  | Clionaidae | Sphaciospongia vagabunda   | 1   | 1   | 1   | 0  | 0   | 0    | 0.068         | 0.068      | 0.068        | 0.068 γ     | γ      |
| 42 | Demospongiae | Clionaidae  | Clionaidae | Sphaciospongia cf. vagabunda | 0   | 0   | 1   | 0  | 0   | 0    | 0.068         | 0.035      | 0.00         | 0.00 γ     | γ      |
| 43 | Demospongiae | Dendroceratida | Darwinellidae | Dendrilla sp.EG1        | 0   | 0   | 1   | 0  | 0   | 0    | 0.205         | 0.205      | 0.023        | 0.023 γ     | γ      |
| 44 | Demospongiae | Dendroceratida | Dictyodendrillidae | Acanthodendrilla cf. sp. Ng1 | 0   | 0   | 0   | 0  | 0   | 0    | 0.023         | 0.023      | 0.00         | 0.00 α     | α      |
| 45 | Demospongiae | Dendroceratida | Dictyodendrillidae | Dictyodendrilla sp. 1 | 0   | 0   | 1   | 0  | 0   | 0    | 0.00          | 0.00       | 0.045        | 0.045 γ     | γ      |
| 46 | Demospongiae | Dendroceratida | Dictyodendrillidae | Dictyodendrilla sp. 1    | 0   | 0   | 0   | 0  | 0   | 0    | 0.068         | 0.068      | 0.00         | 0.00 α     | α      |
| 47 | Demospongiae | Dendroceratida | Dictyodendrillidae | Dictyodendrilla sp. SS1   | 0   | 0   | 0   | 0  | 0   | 0    | 0.00          | 0.00       | 0.023        | 0.023 α     | α      |
| 48 | Demospongiae | Dictyoceratida | Dysideidae | Dysidea sp. 3              | 0   | 0   | 1   | 0  | 0   | 0    | 0.114         | 0.114      | 0.114        | 0.091 γ     | γ      |
| 49 | Demospongiae | Dictyoceratida | Dysideidae | Eurypongia cf. delicatula | 0   | 0   | 0   | 0  | 0   | 0    | 0.023         | 0.023      | 0.00         | 0.00 α     | α      |
| 50 | Demospongiae | Dictyoceratida | Iriniidae | Irinia irregularis         | 0   | 0   | 1   | 0  | 0   | 0    | 0.00          | 0.00       | 0.023        | 0.023 γ     | γ      |
| 51 | Demospongiae | Dictyoceratida | Iriniidae | Irinia cf. irregularis     | 0   | 0   | 0   | 0  | 0   | 0    | 0.023         | 0.023      | 0.00         | 0.00 α     | α      |
| 52 | Demospongiae | Dictyoceratida | Iriniidae | Irinia sp. 1               | 0   | 1   | 1   | 0  | 0   | 0    | 0.318         | 0.169      | 0.364        | 0.132 γ     | γ      |
| 53 | Demospongiae | Dictyoceratida | Iriniidae | Psammocinia bulbosa        | 1   | 0   | 0   | 0  | 0   | 0    | 0.00          | 0.00       | 0.114        | 0.091 β     | β      |
| 54 | Demospongiae | Dictyoceratida | Iriniidae | Psammocinia cf. bulbosa    | 0   | 0   | 1   | 0  | 1   | 0    | 1.045         | 0.363      | 0.00         | 0.00 γ     | γ      |
| 55 | Demospongiae | Dictyoceratida | Iriniidae | Psammocinia sp. 4          | 0   | 0   | 1   | 0  | 0   | 0    | 0.023         | 0.023      | 0.00         | 0.00 γ     | γ      |
| 56 | Demospongiae | Dictyoceratida | Iriniidae | Sarcotragus sp. 2          | 0   | 0   | 1   | 0  | 0   | 0    | 0.023         | 0.023      | 0.00         | 0.00 γ     | γ      |
| 57 | Demospongiae | Dictyoceratida | Iriniidae | Sarcotragus cf. sp. SS7    | 0   | 0   | 0   | 0  | 0   | 0    | 0.023         | 0.023      | 0.00         | 0.00 α     | α      |
| 58 | Demospongiae | Dictyoceratida | Iriniidae | Sarcotragus sp. SS8        | 0   | 1   | 1   | 0  | 1   | 0    | 0.00          | 0.00       | 0.045        | 0.045 γ     | γ      |
| 59 | Demospongiae | Dictyoceratida | Iriniidae | Sarcotragus cf. sp. SS8    | 0   | 0   | 0   | 0  | 0   | 0    | 0.068         | 0.035      | 0.00         | 0.00 α     | α      |
| 60 | Demospongiae | Dictyoceratida | Iriniidae | Sarcotragus sp. PB1        | 0   | 0   | 0   | 0  | 0   | 0    | 0.386         | 0.136      | 0.00         | 0.00 α     | α      |
| #  | Class            | Order      | Family         | Species                                      | NMS | PDO | PIN | NWP | CWI | Mean density | SE density | Mean density | SE density | Record |
|----|------------------|------------|----------------|----------------------------------------------|-----|-----|-----|-----|-----|--------------|------------|--------------|------------|--------|
| 61 | Demospongiae     | Dictyoceratida | Spongiidae     | *Dictyoceratida*                            |     |     |     |     |     |              |            |              |            | α      |
| 62 | Demospongiae     | Dictyoceratida | Spongiidae     | *Spongiidae*                                |     |     |     |     |     |              |            |              |            | γ      |
| 63 | Demospongiae     | Dictyoceratida | Spongiidae     | *Hyattella* cf. *intestinalis*               |     |     |     |     |     |              |            |              |            | α      |
| 64 | Demospongiae     | Dictyoceratida | Spongiidae     | *Rhophaloides*? sp. PB1                     |     |     |     |     |     |              |            |              |            | α      |
| 65 | Demospongiae     | Dictyoceratida | Spongiidae     | *Spongia* (Heterofibria) cf. sp. 2          |     |     |     |     |     |              |            |              |            | α      |
| 66 | Demospongiae     | Dictyoceratida | Spongiidae     | *Spongia* (Spongia) sp. PB1                 |     |     |     |     |     |              |            |              |            | α      |
| 67 | Demospongiae     | Dictyoceratida | Theorectidae   | *Aplysinopsis* sp. 1                        |     |     |     |     |     |              |            |              |            | α      |
| 68 | Demospongiae     | Dictyoceratida | Theorectidae   | *Cacospongia* sp. PB1                       |     |     |     |     |     |              |            |              |            | γ      |
| 69 | Demospongiae     | Dictyoceratida | Theorectidae   | *Cacospongia* sp. PB2                       |     |     |     |     |     |              |            |              |            | α      |
| 70 | Demospongiae     | Dictyoceratida | Theorectidae   | *Cacospongia* sp. SS4                       |     |     |     |     |     |              |            |              |            | β      |
| 71 | Demospongiae     | Dictyoceratida | Theorectidae   | *Hyrtoos* sp. cf. Ng1                       |     |     |     |     |     |              |            |              |            | α      |
| 72 | Demospongiae     | Dictyoceratida | Theorectidae   | *Hyrtoos* sp. PB1                           |     |     |     |     |     |              |            |              |            | α      |
| 73 | Demospongiae     | Dictyoceratida | Theorectidae   | *Hyrtoos* sp. PB2                           |     |     |     |     |     |              |            |              |            | α      |
| 74 | Demospongiae     | Dictyoceratida | Theorectidae   | *Luffariella* sp. PB1                       |     |     |     |     |     |              |            |              |            | α      |
| 75 | Demospongiae     | Haplosclerida | Callyspongiida | *Callyspongia* (Callyspongia) sp. EG3       |     |     |     |     |     |              |            |              |            | γ      |
| 76 | Demospongiae     | Haplosclerida | Callyspongiida | *Callyspongia* (Toxochalina) sp. 1         |     |     |     |     |     |              |            |              |            | γ      |
| 77 | Demospongiae     | Haplosclerida | Callyspongiida | *Callyspongia* cf. sp. EG3                  |     |     |     |     |     |              |            |              |            | γ      |
| 78 | Demospongiae     | Haplosclerida | Chalinidae     | *Halicina* (Gellius) ambionensis             |     |     |     |     |     |              |            |              |            | γ      |
| 79 | Demospongiae     | Haplosclerida | Chalinidae     | *Halicina* (Halicina) cf. sp. Ng2           |     |     |     |     |     |              |            |              |            | γ      |
| 80 | Demospongiae     | Haplosclerida | Chalinidae     | *Halicina* (Halicina) sp. PB1               |     |     |     |     |     |              |            |              |            | γ      |
| 81 | Demospongiae     | Haplosclerida | Chalinidae     | *Halicina* sp. 10                           |     |     |     |     |     |              |            |              |            | γ      |
| 82 | Demospongiae     | Haplosclerida | Chalinidae     | *Halicina* cf. sp. 17                       |     |     |     |     |     |              |            |              |            | γ      |
| 83 | Demospongiae     | Haplosclerida | Niphatidae     | *Amphimedon lamellata*                      |     |     |     |     |     |              |            |              |            | γ      |
| 84 | Demospongiae     | Haplosclerida | Niphatidae     | *Amphimedon paraviridis*                    |     |     |     |     |     |              |            |              |            | γ      |
| 85 | Demospongiae     | Haplosclerida | Petrosiidae    | *Nepetrosis exigua*                         |     |     |     |     |     |              |            |              |            | γ      |
| 86 | Demospongiae     | Haplosclerida | Petrosiidae    | *Petrosia* (Petrosia) sp. SS4               |     |     |     |     |     |              |            |              |            | γ      |
| 87 | Demospongiae     | Haplosclerida | Petrosiidae    | *Xestospongia* sp. 1                        |     |     |     |     |     |              |            |              |            | γ      |
| 88 | Demospongiae     | Haplosclerida | Petrosiidae    | *Xestospongia* sp. 3                        |     |     |     |     |     |              |            |              |            | γ      |
| 89 | Demospongiae     | Haplosclerida | Petrosiidae    | *Xestospongia* cf. sp. 4                    |     |     |     |     |     |              |            |              |            | γ      |
| 90 | Demospongiae     | Haplosclerida | Petrosiidae    | *Xestospongia* cf. sp. 4                    |     |     |     |     |     |              |            |              |            | γ      |
| #  | Class          | Order         | Family            | Species              | NMS | PHI | PIN | NIN | NWP | CMT | Mean density | SE density | Onslow pre-dredging | Onslow post-dredging |
|----|----------------|---------------|-------------------|----------------------|-----|-----|-----|-----|-----|-----|--------------|-------------|----------------------|---------------------|
| 91 | Demospongiae   | Haplosclerida | Petrosiidae       | Xestospongia sp. 5   | 0   | 1   | 1   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 92 | Demospongiae   | Haplosclerida | Petrosiidae       | Xestospongia sp. P2  | 0   | 0   | 0   | 0   | 0   | 0   | 0.182        | 0.112       | 0                    | 0                    |
| 93 | Demospongiae   | Haplosclerida | Petrosiidae       | Xestospongia sp. PB1 | 0   | 0   | 0   | 0   | 0   | 0   | 0.182        | 0.112       | 0                    | 0                    |
| 94 | Demospongiae   | Haplosclerida | Petrosiidae       | Xestospongia testudinaria | 1   | 1   | 1   | 0   | 0   | 0   | 0.273        | 0.152       | 0.045                | 0.045               |
| 95 | Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia CERF sp. 2 | 0   | 0   | 0   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 96 | Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia CERF cf. sp. 2 | 0   | 0   | 0   | 0   | 0   | 0   | 0.045        | 0.045       | 0                    | 0                    |
| 97 | Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia sp. 2      | 0   | 0   | 1   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 98 | Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia sp. 5      | 0   | 0   | 1   | 0   | 0   | 0   | 0.682        | 0.188       | 0.023                | 0.023               |
| 99 | Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia cf. sp. 7  | 0   | 0   | 0   | 0   | 0   | 0   | 0.864        | 0.353       | 0.091                | 0.091               |
| 100| Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia cf. sp. 8  | 0   | 0   | 0   | 0   | 0   | 0   | 0.227        | 0.227       | 0                    | 0                    |
| 101| Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia sp. SS13   | 0   | 1   | 0   | 1   | 0   | 0   | 0.636        | 0.365       | 0.045                | 0.045               |
| 102| Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia cf. sp. SS2| 0   | 0   | 0   | 0   | 0   | 0   | 0.0227       | 0.023       | 0                    | 0                    |
| 103| Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia sp. SS9    | 0   | 0   | 0   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 104| Demospongiae   | Haplosclerida | Phloeodictyidae   | Oceanapia sp.7       | 0   | 0   | 1   | 0   | 0   | 0   | 0.432        | 0.327       | 0                    | 0                    |
| 105| Demospongiae   | Haplosclerida | Phloeodictyidae   | Siphonodictyon sp. SS9| 0   | 0   | 0   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 106| Demospongiae   | Poecilosclerida| Chondropsidae     | Chondropsis sp.1     | 0   | 0   | 1   | 0   | 0   | 0   | 0.068        | 0.049       | 0                    | 0                    |
| 107| Demospongiae   | Poecilosclerida| Coelosphaeridae   | Coelosphaera (Coelosphaera) sp. PB1 | 0   | 0   | 0   | 0   | 0   | 0   | 0.068        | 0.068       | 0                    | 0                    |
| 108| Demospongiae   | Poecilosclerida| Crambeidae        | Monanchora sp. 7     | 0   | 0   | 0   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 109| Demospongiae   | Poecilosclerida| Hypodoidesmidae   | Hamigera sp. PB1     | 0   | 0   | 0   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 110| Demospongiae   | Poecilosclerida| Iotrochotidae     | Iotrochota sp. KMB1  | 0   | 0   | 0   | 0   | 0   | 0   | 0.114        | 0.070       | 0.045                | 0.045               |
| 111| Demospongiae   | Poecilosclerida| Iotrochotidae     | Iotrochota sp. KMB2  | 0   | 0   | 0   | 0   | 0   | 0   | 0.045        | 0.045       | 0                    | 0                    |
| 112| Demospongiae   | Poecilosclerida| Microcionidae     | Clathria (Thalysias) abietina | 1   | 1   | 1   | 1   | 0   | 0   | 0.182        | 0.112       | 0.023                | 0.023               |
| 113| Demospongiae   | Poecilosclerida| Microcionidae     | Clathria (Thalysias) lendenfeldi | 1   | 1   | 1   | 1   | 0   | 0   | 0.000        | 0.000       | 0.386                | 0.242               |
| 114| Demospongiae   | Poecilosclerida| Microcionidae     | Clathria (Thalysias) cf. lendenfeldi | 0   | 0   | 0   | 0   | 0   | 0   | 0.114        | 0.062       | 0                    | 0                    |
| 115| Demospongiae   | Poecilosclerida| Microcionidae     | Clathria (Thalysias) major | 1   | 1   | 1   | 0   | 0   | 0   | 0.364        | 0.203       | 0                    | 0                    |
| 116| Demospongiae   | Poecilosclerida| Microcionidae     | Clathria (Thalysias) reinwardti | 0   | 0   | 1   | 1   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 117| Demospongiae   | Poecilosclerida| Microcionidae     | Clathria (Thalysias) vulpina | 1   | 1   | 1   | 0   | 0   | 0   | 0.068        | 0.035       | 0.045                | 0.045               |
| 118| Demospongiae   | Poecilosclerida| Mycalidae         | Mycale (Aegopogon) sp. 2 | 0   | 0   | 1   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 119| Demospongiae   | Poecilosclerida| Mycalidae         | Mycale (Aegopogon) sp. PB1 | 0   | 0   | 0   | 0   | 0   | 0   | 0.023        | 0.023       | 0                    | 0                    |
| 120| Demospongiae   | Poecilosclerida| Mycalidae         | Mycale (Arenochalina) mirabilis | 1   | 0   | 0   | 0   | 0   | 0   | 0.432        | 0.122       | 0.068                | 0.068               |
### Supplementary Table 4 continued – pg. 5

| #   | Class          | Order          | Family | Species                        | NWS | PIO | PIN | MIN | NMP | CWT | Mean density | SE density | Onslow pre-dredging | Mean density | SE density | Onslow post-dredging | Mean density | SE density | Record |
|-----|----------------|----------------|--------|--------------------------------|-----|-----|-----|-----|-----|-----|--------------|-------------|----------------------|--------------|-------------|----------------------|--------------|-------------|--------|
| 121 | Demospongiae   | Poecilosclerida | Mycalidae | Mycale (Arenochalina) cf. mirabilis | 0   | 0   | 0   | 0   | 0   | 0.023 | 0.023         |             | 0         | 0               | α            |             |         |
| 122 | Demospongiae   | Poecilosclerida | Mycalidae | Mycale (Carmia) sp. 5          | 0   | 0   | 1   | 0   | 0   | 0.068 | 0.068         | 0.523       | 0.450     | y          |              |             |             |        |
| 123 | Demospongiae   | Poecilosclerida | Mycalidae | Mycale (Grapelio) sp. PB1       | 0   | 0   | 0   | 0   | 0   | 0     | 0.023         | 0.023       | α          | α          |              |             |             |        |
| 124 | Demospongiae   | Polymastiida    | Polymastiidae | Polymastia sp. PB1             | 0   | 0   | 0   | 0   | 0   | 0     | 0.068         | 0.068       | α          | α          |              |             |             |        |
| 125 | Demospongiae   | Polymastiida    | Polymastiidae | Polymastia sp. PB2             | 0   | 0   | 0   | 0   | 0   | 0     | 0.045         | 0.045       | α          | α          |              |             |             |        |
| 126 | Demospongiae   | Polymastiida    | Polymastiidae | Porphyria sp. PB1              | 0   | 0   | 0   | 0   | 0   | 0     | 0.230         | 0.230       | α          | α          |              |             |             |        |
| 127 | Demospongiae   | Scopalinida     | Scopalinidae | Styliisca flabelliformis       | 0   | 1   | 0   | 0   | 0   | 0.159 | 0.091         | 0.091       | 0.061     | y          |              |             |             |        |
| 128 | Demospongiae   | Scopolinida     | Scopalinidae | Styliisca cf. flabelliformis   | 0   | 0   | 0   | 0   | 0   | 0.091 | 0.070         |             | 0         | 0               | α            |             |         |
| 129 | Demospongiae   | Suberitida      | Halichondriidae | Ciocalypta sp. PB1            | 0   | 0   | 0   | 0   | 0   | 0     | 0.159         | 0.136       | α          | α          |              |             |             |        |
| 130 | Demospongiae   | Suberitida      | Halichondriidae | Ciocalypta sp. PB2            | 0   | 0   | 0   | 0   | 0   | 0     | 0.045         | 0.045       | α          | α          |              |             |             |        |
| 131 | Demospongiae   | Suberitida      | Halichondriidae | Ciocalypta sp. PB3            | 0   | 0   | 0   | 0   | 0   | 0     | 0.023         | 0.023       | α          | α          |              |             |             |        |
| 132 | Demospongiae   | Suberitida      | Halichondriidae | Ciocalypta tyleri             | 0   | 0   | 0   | 0   | 0   | 0     | 0.159         | 0.113       | y          | y          |              |             |             |        |
| 133 | Demospongiae   | Suberitida      | Halichondriidae | Ciocalypta cf. tyleri         | 0   | 0   | 0   | 0   | 0   | 0     | 0.091         | 0.091       | α          | α          |              |             |             |        |
| 134 | Demospongiae   | Suberitida      | Halichondriidae | Halichondra sp. PB1          | 0   | 0   | 0   | 0   | 0   | 0     | 0.227         | 0.156       | α          | α          |              |             |             |        |
| 135 | Demospongiae   | Suberitida      | Suberitidae    | Aaptos sp. KMB1               | 0   | 0   | 0   | 0   | 0   | 0     | 0.023         | 0.023       | α          | α          |              |             |             |        |
| 136 | Demospongiae   | Tethyida        | Hemisterellidae | Axos cliftoni                | 1   | 1   | 1   | 1   | 0   | 0     | 0.045         | 0.045       | y          | y          |              |             |             |        |
| 137 | Demospongiae   | Tethyida        | Hemisterellidae | Axos flabelliformis         | 1   | 1   | 1   | 0   | 0   | 0     | 0.455         | 0.138       | y          | y          |              |             |             |        |
| 138 | Demospongiae   | Tetractinellida | Ancorinidae   | Jaspis sp. SS4 cf.           | 0   | 0   | 0   | 0   | 0   | 0     | 0.045         | 0.045       | α          | α          |              |             |             |        |
| 139 | Demospongiae   | Tetractinellida | Ancorinidae   | Rhodactrellia cf. globostellata | 0   | 0   | 0   | 0   | 0   | 0     | 0.273         | 0.137       | α          | α          |              |             |             |        |
| 140 | Demospongiae   | Tetractinellida | Tetillidae    | Cinachyra sp. BB21           | 0   | 0   | 0   | 0   | 0   | 0     | 0.477         | 0.180       | α          | α          |              |             |             |        |
| 141 | Demospongiae   | Tetractinellida | Tetillidae    | Cinachyrella cf. australiensis | 0   | 1   | 1   | 1   | 0   | 1     | 0.136         | 0.062       | y          | y          |              |             |             |        |
| 142 | Demospongiae   | Verongiida      | Aplysinidae   | Aplysina cf. sp. 3           | 0   | 0   | 0   | 0   | 0   | 0     | 0.114         | 0.114       | α          | α          |              |             |             |        |
| 143 | Demospongiae   | Verongiida      | Iantheilidae  | Iantheilla basta           | 1   | 1   | 1   | 0   | 1   | 0     | 0.386         | 0.192       | α          | α          |              |             |             |        |
| 144 | Demospongiae   | Verongiida      | Iantheilidae  | Iantheilla flabelliformis    | 1   | 1   | 1   | 0   | 0   | 0     | 0.659         | 0.266       | y          | y          |              |             |             |        |
| 145 | Demospongiae   | Verongiida      | Iantheilidae  | Iantheilla cf. flabelliformis | 0   | 0   | 1   | 1   | 0   | 0     | 0.023         | 0.023       | y          | y          |              |             |             |        |
| 146 | Demospongiae   | Verongiida      | Iantheilidae  | Iantheilla cf. labyrinthus   | 0   | 0   | 1   | 0   | 0   | 0     | 0.045         | 0.030       | y          | y          |              |             |             |        |
| 147 | Demospongiae   | Verongiida      | Pseudoceratinida | Pseudoceratinis sp. 2        | 0   | 0   | 1   | 0   | 0   | 0     | 0.864         | 0.211       | y          | y          |              |             |             |        |
| 148 | Demospongiae   | Verongiida      | Pseudoceratinida | Pseudoceratina verrucosa    | 0   | 0   | 0   | 0   | 0   | 0     | 0.159         | 0.136       | α          | α          |              |             |             |        |
| 149 | Demospongiae   | Verongiida      | Pseudoceratinida | Pseudoceratina cf. verrucosa | 0   | 0   | 1   | 0   | 0   | 0     | 0.068         | 0.049       | y          | y          |              |             |             |        |
| 150 | Homoscleromorpha | Homosclerosphorida | Plakinidae | Plakortis sp. PB1 | 0   | 0   | 0   | 0   | 0   | 0     | 0.136         | 0.136       | α          | α          |              |             |             |        |
Supplementary Figure 1: Nephelometric turbidity units (NTUs, top graphs) and daily light integral (DLI, bottom graphs) collected every 30 minutes at the sixteen sites. Site name and corresponding site number are provided at the top left of the figure. Figures on the left hand side represent maximum daily NTU and maximum daily DLI recorded for the sites two years before dredging started (17 May 2011 to 10 April 2013) and during the two year dredging operation (11 April 2013 to 27 February 2015). Blue vertical lines represent time when dredging started. Grey vertical bars with red dashed vertical lines represent the duration and start date of the 5 cyclones that passed near the study area respectively. The middle figures show the 80th, 95th, 99th and 100th (maximum) percentiles of NTUs, and the 20th, 10th, 5th and 0th (minimum) over a range of running means intervals (from 1 hour to 30 days) before (dashed lines) and during the dredging (solid lines), excluding periods of cyclones. Figures on the right represent cumulative probability curves for NTU and DLI, before dredging (dashed lines) and during dredging (solid lines), with area between the before and during dredging curves representing exceedance (for NTU) and reduction (for DLI).
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Supplementary Figure 1 continued - pg 9

**Supplementary Figure 1**: Nephelometric turbidity units (NTUs, top graphs) and daily light integral (DLI, bottom graphs) collected every 30 minutes at the sixteen sites. Site name and corresponding site number are provided at the top left of the figure. Figures on the left hand side represent maximum daily NTU and maximum daily DLI recorded for the sites two years before dredging started (17 May 2011 to 10 April 2013) and during the two year dredging operation (11 April 2013 to 27 February 2015). Blue vertical lines represent time when dredging started. Grey vertical bars with red dashed vertical lines represent the duration and start date of the 5 cyclones that passed near the study area respectively. The middle figures show the 80th, 95th, 99th and 100th (maximum) percentiles of NTUs, and the 20th, 10th, 5th and 0th (minimum) over a range of running means intervals (from 1 hour to 30 days) before (dashed lines) and during the dredging (solid lines), excluding periods of cyclones. Figures on the right represent cumulative probability curves for NTU and DLI, before dredging (dashed lines) and during dredging (solid lines), with area between the before and during dredging curves representing exceedance (for NTU) and reduction (for DLI).
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Supplementary Figure 1: Nephelometric turbidity units (NTUs, top graphs) and daily light integral (DLI, bottom graphs) collected every 30 minutes at the sixteen sites. Site name and corresponding site number are provided at the top left of the figure. Figures on the left hand side represent maximum daily NTU and maximum daily DLI recorded for the sites two years before dredging started (17 May 2011 to 10 April 2013) and during the two year dredging operation (11 April 2013 to 27 February 2015). Blue vertical lines represent time when dredging started. Grey vertical bars with red dashed vertical lines represent the duration and start date of the 5 cyclones that passed near the study area respectively. The middle figures show the 80th, 95th, 99th and 100th (maximum) percentiles of NTUs, and the 20th, 10th, 5th and 0th (minimum) over a range of running means intervals (from 1 hour to 30 days) before (dashed lines) and during the dredging (solid lines), excluding periods of cyclones. Figures on the right represent cumulative probability curves for NTU and DLI, before dredging (dashed lines) and during dredging (solid lines), with area between the before and during dredging curves representing exceedance (for NTU) and reduction (for DLI).
Supplementary Figure 2: Cluster analysis of pre-dredging NTUs at the 16 water monitoring sites (refer to Table 1 for full site names). Complete linkage dendrogram of log transformed 80th percentile moving averages across multiple running mean intervals (1h to 30 days) for the pre-dredging period ($n_{days}$ = ca. 690).
Supplementary Figure 3: A) Map showing the mid-points of the 6 towed video transects closest (within 700 m; Tow 10, 12, 13, 14, 15 and 27) to the End of Channel water monitoring site (specific mid-point distance to the edge of the channel are shown in parentheses under the Tow number). B) Taxa and C) sponge functional morphologies densities (individuals m$^{-2}$) for the individual towed video transects in closest proximity to the End of Channel water monitoring site.
Supplementary Figure 4: Summary figures of means (±SE), and multivariate and univariate PERMANOVA stats for A) broad taxonomic benthic community categories and B) sponge functional morphologies for the 6 towed video transects in proximity to the End of Channel water monitoring site (see Supplementary Figure 3 for map and details for individual transect). Texts in red represent significant results and text in blue represent results which minimally exceeded p = 0.05.

**PERMANOVA table of results: Multivariate**

| Source | df  | SS   | MS   | Pseudo-F | P(perm) | Unique perm |
|--------|-----|------|------|-----------|---------|-------------|
| Dr     | 1   | 1502 | 1502 | 2.0381    | 0.1086  | 462         |
| Res    | 10  | 7369.3 | 736.93 | 0.0456 | 0.0046007 | 0.9563 | 32 |
| Total  | 11  | 8871.3 |  |

**PERMANOVA table of results: Univariate**

| Taxa       | SS   | MS   | Pseudo-F | P(perm) | Unique perm |
|------------|------|------|----------|---------|-------------|
| Asc_Col    | 3500.9 | 3500.9 | 3.7076 | 0.0506 | 336 |
| Asc_Sol    | 2530 | 2530 | 1.4265 | 0.3101 | 8 |
| Gong Colonial | 167.59 | 167.59 | 0.5285 | 0.5084 | 336 |
| Hydro      | 6641 | 6641 | 12.094 | 0.0143 | 210 |
| Bryo       | 172.84 | 172.84 | 0.3119 | 0.8068 | 462 |
| Macroalgae | 3.51 | 3.51 | 0.0046007 | 0.9563 | 32 |
| Hard coral | 1117.2 | 1117.2 | 7.1537 | 0.0276 | 462 |
| Soft coral | 14.589 | 14.589 | 0.0319 | 0.841 | 8 |
| Sponges    | 26.122 | 26.122 | 0.068136 | 0.8917 | 462 |
| Starfish   | 0.50646 | 0.50646 | 0.0020177 | 1 | 2 |
| Seagrass   | 204.27 | 204.27 | 1 | 1 | 1 |
| Nudibranch | 0.94099 | 0.94099 | 0.0036409 | 1 | 2 |
| Polychaete | 869.59 | 869.59 | 2.2141 | 0.457 | 2 |
| Zoanthid   | 587.38 | 587.38 | 1.1082 | 0.5562 | 8 |
| Crinoid    | No test | No test | No test | No test | No test |
| Echinoderm | No test | No test | No test | No test | No test |
| Holothurian | No test | No test | No test | No test | No test |

**PERMANOVA table of results: Multivariate**

| Source | df  | SS   | MS   | Pseudo-F | P(perm) | Unique perm |
|--------|-----|------|------|----------|---------|-------------|
| Dr     | 1   | 491.26 | 491.26 | 0.41104 | 0.7178 | 462 |
| Res    | 10  | 11952 | 1195.2 | 0.0456 | 0.0046007 | 0.9563 | 32 |
| Total  | 11  | 12443 |  |

**PERMANOVA table of results: Univariate**

| Morph       | SS   | MS   | Pseudo-F | P(perm) | Unique perm |
|-------------|------|------|----------|---------|-------------|
| Creeping (EN-cg) | 58.846 | 58.846 | 0.059604 | 0.7382 | 32 |
| Endolithic (EN-en) | 137.31 | 137.31 | 0.16434 | 0.6992 | 32 |
| Crust (EN-cr) | 231.55 | 231.55 | 2.2247 | 0.1751 | 462 |
| Simple (M-s) | 136.91 | 136.91 | 0.52744 | 0.4638 | 462 |
| Ball (M-bl) | 76.208 | 76.208 | 0.17394 | 0.8433 | 462 |
| Cryptic (M-crp) | 572.84 | 572.84 | 0.76778 | 0.4375 | 336 |
| Tabulate (C-tab) | 599.36 | 599.36 | 1.1332 | 0.5396 | 8 |
| Incomplete (C-inc) | 1085.1 | 1085.1 | 3.0605 | 0.2389 | 16 |
| Wide (C-wd) | 663.44 | 663.44 | 1.4827 | 0.2415 | 32 |
| Narrow (C-nw) | 118.73 | 118.73 | 1 | 1 | 1 |
| Laminar (E-lam) | 120.95 | 120.95 | 1 | 1 | 1 |
| Palmate (E-pal) | 48.501 | 48.501 | 0.006059 | 0.8982 | 336 |
| Branching (E-br) | 17.44 | 17.44 | 0.053995 | 1 | 2 |
| Simple (E-s) | 300.86 | 300.86 | 0.29229 | 0.7439 | 210 |
| Stalked (E-st) | 275.23 | 275.23 | 0.40935 | 0.5203 | 210 |

**PERMANOVA table of results: Univariate**

| Taxa       | SS   | MS   | Pseudo-F | P(perm) | Unique perm |
|------------|------|------|----------|---------|-------------|
| Asc_Col    | 3500.9 | 3500.9 | 3.7076 | 0.0506 | 336 |
| Asc_Sol    | 2530 | 2530 | 1.4265 | 0.3101 | 8 |
| Gong Colonial | 167.59 | 167.59 | 0.5285 | 0.5084 | 336 |
| Hydro      | 6641 | 6641 | 12.094 | 0.0143 | 210 |
| Bryo       | 172.84 | 172.84 | 0.3119 | 0.8068 | 462 |
| Macroalgae | 3.51 | 3.51 | 0.0046007 | 0.9563 | 32 |
| Hard coral | 1117.2 | 1117.2 | 7.1537 | 0.0276 | 462 |
| Soft coral | 14.589 | 14.589 | 0.0319 | 0.841 | 8 |
| Sponges    | 26.122 | 26.122 | 0.068136 | 0.8917 | 462 |
| Starfish   | 0.50646 | 0.50646 | 0.0020177 | 1 | 2 |
| Seagrass   | 204.27 | 204.27 | 1 | 1 | 1 |
| Nudibranch | 0.94099 | 0.94099 | 0.0036409 | 1 | 2 |
| Polychaete | 869.59 | 869.59 | 2.2141 | 0.457 | 2 |
| Zoanthid   | 587.38 | 587.38 | 1.1082 | 0.5562 | 8 |
Supplementary Figure 5: Landsat images of the Onslow study area through time. **4 May 2013**: Dredging plume visible within the Inner channel zone with most intense dredging occurring around PSD sampling transect S2. The plume extends beyond the Inner zone north and west. Ashburton River discharge not visible. **24 August 2013**: Dredging plume visible within Inner zone and trunkline west of channel. Sediment plume from the spoil ground presumably from disposal of dredge sediment migrates westwards towards Thevenard Island. Large discharge from the Ashburton River migrating eastwards close to shore and potentially migrating over the ROLLER water monitoring site and the S1 PSD sampling transect. **25 September 2013**: Turbidity appeared to be affecting most of the study area, with dredging related plume visible within the Inner, Mid and Outer zone. Sediment plume from the spoil ground is moving eastwards. Coastal area from the Ashburton River mouth past Onslow is highly turbid, presumably from eastwards/north-eastwards migration of the riverine runoff plume and/or re-suspension of sediment in shallow coastal water. The BESS water monitoring site was not affected by sediment plumes in all images. Landsat images produced by M. Broomhall and P. Fears, Remote Sensing and Satellite Research Group, Curtin University. Source of Landsat data: US Geological Survey.
Supplementary Figure 5 (continued): Landsat images of the Onslow study area through time (continued). 4 March 2014: Dredging plume visible within the Outer channel zone in proximity to the ENDCH water monitoring site and S7 PSD sampling transect. Sediment plume from the spoil ground appears to migrate outwards from the centre of the spoil zone. The Ashburton River discharge is visible, however, is less intense compared to that observed on 24 August 2013. 23 May 2014: Dredging plume visible within the Inner zone in proximity to PSD sampling transect S2, S3 and S4. A large riverine plume from the Ashburton River is visible, migrating north and eastwards towards the ROLLER water monitoring site. 25 September 2014: Dredging plume visible in all channel zones, with most prominent plume in the Mid zone. Sediment plume from the spoil ground is less intense than the previous images. Only minor riverine discharge can be observed from the Ashburton River. The BESS water monitoring site was not affected by sediment plumes in all images. Landsat images produced by M. Broomhall and P. Fears, Remote Sensing and Satellite Research Group, Curtin University. Source of Landsat data: US Geological Survey.
Appendices

Appendix 1. Assessment Of Macroalgae, Seagrass And Filter Feeders Cover Between A Reference Site Where Water Quality Was Not Affected By Dredging (Bessieres Island), And Sites Where Water Quality Was Affected By Dredging (Outer Channel Zone)

Macroalgae exhibited the highest reduction in abundance (-61%), followed by rhodoliths (-27%), in the before and after surveys described in the main body of this report. We concluded that this may be due to (1) dredging related pressures (light reduction etc.), (2) the effects of cyclone Olwyn, and (3) seasonal patterns of abundance since sampling was conducted at a different time of year.

Towards the end of this study, additional information on macroalgae, seagrass and filter feeders cover before and after the Wheatstone dredging project became available from the dredging proponents from their compliance monitoring programs. The much more widespread survey information which included sampling before and after dredging, and at the same time of year, afforded an opportunity to examine seasonal patterns in greater detail. We therefore used the information to see if there were any further insights into the changes in primarily the macroalgae abundances using the compliance monitoring data from a reference site where water quality was not affected by dredging at Bessieres Island, and sites within the Outer Channel Zone where water quality was affected by dredging (see Figure 1 of the Sub-project 6.3 report for a map).

Bessieres Island experienced the lowest turbidity out of all water quality monitoring sites with P80 <1 NTU across all running mean intervals (1 d to 30 d), even during the dredging phase. From this analysis Bessieres Island was considered to be largely unaffected by dredging. In contrast, the P80 of the turbidity values at sites within the Outer Channel Zone (ENDCH, PAROO, SALAD and GORGSW) ranged from 2.4 to 3.3 NTU (over running mean interval from 1 d to 30 d). This is consistent with the site being regularly influenced by the dredging plumes.

Two pre-dredging surveys were performed by the proponent, one in September 2012 (Pre 1) and one in December 2012 (Pre 2; Figure A1). One post-dredging survey was performed in December 2014 (Post), before the passing of cyclone Olwyn (in March 2015). A remotely operated underwater vehicle (ROV) was used to capture downward facing images of the benthos at 6 survey cells (500 m × 500 m cell; 5 × 100 m transects per cell) at the Reference site and 9 cells from the Outer Channel Zone. Images were scored for macroalgae, seagrass and filter feeders using a 5-point intercept protocol in the program CPCe (Kohler and Gill 2006). Averaged cover data (by cells, and within site [Reference and Outer Channel Zone] and dredging periods [Pre 1, Pre 2 and Post]) were square root transformed and analysed in the program PRIMER/ PERMANOVA+ Version 7, using the Bray-Curtis similarity matrix and 9999 permutations.

At the pre-dredging surveys, the Reference site showed higher cover of macroalgae (up to 1.9× higher) and filter feeders (up to 5.3× higher), with concomitant lower cover of seagrass (up to 0.13× lower; Figure A1, Table A1 and A2A). There was a significant effect of sites (Reference and Outer Channel Zone) and dredging period (Pre 1, Pre 2 and Post) on the cover of benthic communities, specifically macroalgae, seagrass and filter feeders (PERMANOVA: Location: Pseudo-F = 6.65, p = 0.0003; Dredging period: Pseudo-F = 3.00, p = 0.0053). However, there were no significant interactions between sites and dredging periods, which indicate that dredging was not a significant driver of the observed differences in the cover benthic community cover.
Comparisons of benthic filter feeder communities before and after a large-scale capital dredging program

Figure A1: Mean proportion cover (% ± SE) of A) macroalgae, B) seagrass and C) combined filter feeder at Bessieres Island (Reference site; where water quality was not affected by dredging) and Outer Channel Zone (where water quality was affected by dredging) at two pre-dredging surveys (Pre 1: September 2012 and Pre 2: December 2012) and a post-dredging survey (Post: December 2014). Vertical grey dashed line demarcates the Reference site to the left and the Outer Channel Zone to the right. The before and during dredging daily maximum nephelometer turbidity unit (NTU) plot for D) Bessieres Island (BESS; Reference) and the End of Channel water monitoring site located in the Outer Channel Zone (ENDCH) highlight when dredging began (blue solid vertical line), and when the Pre 1 (green arrow), Pre 2 (blue arrow) and Post (red arrow) surveys were conducted. Grey vertical bars and dashed vertical red lines represent the duration and start date of the 6 cyclones what passed near the study area respectively. In chronological order, the 6 cyclones include Tropical Cyclone Iggy (Category 2), TC Lua (Cat 4), TC Mitchell (Cat 1), TC Narelle (Cat 4), TC Rusty (Cat 4) and TC Christine (Cat 4).

To further assess changes in individual taxa between Reference site and Outer Channel Zone, and at before and after dredging, only Pre 2 and Post data were used. These dredging periods were selected as they were both conducted in the month of December, thus reducing any potential effects of seasonality.

The results showed that while the Outer Channel Zone had lower cover of filter feeders than the Reference site, neither site or dredging period resulted in any detectable change in filter feeder community, highlighting the stability of filter feeder communities through space and time at the study area (Figure A1C; PERMANOVA: Pseudo-F = 1.94 and 0.79 respectively, p > 0.05). This confirms a conclusion of the main body of this report. Dredging periods did not affect the cover of seagrass (PERMANOVA: Pseudo-F = 0.35, p > 0.05), but were different between sites (Figure A1B; PERMANOVA: Pseudo-F = 39.78, p = 0.0001). No significant interaction between sites and dredging periods were detected for filter feeder and seagrass.

According to the proponent’s surveys, macroalgae showed a significant reduction in cover, by 0.2–0.3× at both the Reference site and Outer Channel Zone after dredging (Figure A1A and Table A2B), with no effect of sites (PERMANOVA: Pseudo-F = 1.51, p > 0.05) and an effect of dredging periods on macroalgae detected (PERMANOVA: Pseudo-F = 5.99, p = 0.003). No significant interaction between sites and dredging periods were
Comparisons of benthic filter feeder communities before and after a large-scale capital dredging program detected for macroalgae. Considering the proponent’s surveys were performed during the same period of the year (pre-dredging in Dec 2012, and post-dredging in Dec 2014), and post-dredging surveys were performed prior to the passing of cyclone Olwyn in March 2015, this suggests that factors other than dredging or cyclone Olwyn could have contributed to the reductions of macroalgae at the study area. Of particular note was that there were other cyclones (TC Mitchell, Narelle and Rusty during pre-dredging period, and TC Christine during dredging period; Figure A1D) that occurred within the region between these surveys. Bessieres Island is much more exposed compared to the Outer Channel Zone (which can be protected by Thevenard Island to the north), and the passing of these cyclones, and associated swells could have resulted in the displacement of macroalgae from the benthos at this reference site.

Macroalgae are a highly temporally variable component of benthic communities (Vuki and Price 1994; Lefèvre and Bellwood 2010; Abdul Wahab et al. 2014) and even sampling at the same time of year (i.e. the same month) may not be sufficient to identify inter-annual variability. This additional analysis has not provided any further insights into the underlying cause of the decrease in macroalgal communities before and after dredging described in the report and the conclusions of the main body of the report remain that changes in macroalgal abundance could be due to dredging related pressures (e.g. light reduction) or seasonality but could also be due to the effects of cyclone Olwyn.

Table A1: Mean cover (%) and standard error of macroalgae (MA), seagrass (SG) and filter feeder (FF) at the Reference site (Bessieres Island; where water quality was not affected by dredging) and Outer Channel Zone (where water quality was affected by dredging) at two pre-dredging surveys (Pre 1: September 2012 and Pre 2: December 2012) and a post-dredging survey (Post: December 2014).

| Site                  | Dredge period | n | Macroalgae | Seagrass | Filter Feeders |
|-----------------------|---------------|---|------------|----------|----------------|
|                       |               |   | Mean       | SE       | Mean           | SE   | Mean | SE     |
| Reference             | Pre 1         | 6 | 13.93      | 4.91     | 0.10           | 0.07 | 2.65 | 1.21   |
| Reference             | Pre 2         | 6 | 18.93      | 6.64     | 0.14           | 0.09 | 2.29 | 0.88   |
| Reference             | Post          | 6 | 4.31       | 1.43     | 0.01           | 0.01 | 2.85 | 1.20   |
| Outer Channel Zone    | Pre 1         | 9 | 9.77       | 1.59     | 0.78           | 0.34 | 0.75 | 0.23   |
| Outer Channel Zone    | Pre 2         | 9 | 10.12      | 1.43     | 1.68           | 0.52 | 1.19 | 0.43   |
| Outer Channel Zone    | Post          | 9 | 2.87       | 0.79     | 2.24           | 0.60 | 1.00 | 0.42   |

Table A2: Ratio of mean macroalgae, seagrass and filter feeder cover (%) between A) Reference/ Outer Channel Zone at each dredge survey period (Pre 1, Pre 2 and Post), and B) Post/ Pre 2 for the Reference site and Outer Channel Zone. Post and Pre 2 were used for comparison as they were both sampled in the month of December, eliminating the effects of seasonality. For both tables, positive values represent an increase in cover, and negative values represent a decrease in cover, with a value of 1 representing no change.

| Dredge period | Reference/ Outer Channel Zone | Post/ Pre 2 |
|---------------|-------------------------------|-------------|
|               | MA | SG | FF | MA | SG | FF |
| Pre 1         | 1.43 | 0.13 | 5.29 | 0.23 | 0.07 | 1.35 |
| Pre 2         | 1.87 | 0.08 | 2.05 | 0.28 | 1.34 | 0.97 |
| Post          | 1.50 | 0.00 | 2.85 |
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