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Abstract

2020 Elsevier Ltd Intensification of lateritic nickel mining in Southeast Asia and Melanesia potentially threatens coastal ecosystems from increased exposure to nickel and suspended sediment. This study investigated the response of *Acropora muricata* when exposed to either dissolved nickel, clean suspended sediment or nickel-contaminated suspended sediment for 7 days, followed by a 7-d recovery period. Significant bleaching and accumulation of nickel in coral tissue was observed only after exposure to high dissolved nickel concentrations and nickel-spiked suspended sediment. No effect on *A. muricata* was observed from exposure to a particulate-bound nickel concentration of 60 mg/kg acid-extractable nickel at a suspended sediment concentration of 30 mg/L TSS. This study demonstrates that bioavailability of nickel associated with suspended sediment exposure plays a key role in influencing nickel toxicity to corals. These findings assist in assessments of risk posed by increasing nickel mining activities on tropical marine ecosystems.

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Effects of dissolved nickel and nickel-contaminated suspended sediment on the scleractinian coral, *Acropora muricata*

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Abstract

Intensification of lateritic nickel mining in Southeast Asia and Melanesia potentially threatens coastal ecosystems from increased exposure to nickel and suspended sediment. This study investigated the response of *Acropora muricata* when exposed to either dissolved nickel, clean suspended sediment or nickel-contaminated suspended sediment for 7 days, followed by a 7-d recovery period.

Significant bleaching and accumulation of nickel in coral tissue was observed only after exposure to high dissolved nickel concentrations and nickel-spiked suspended sediment. No effect on *A. muricata* was observed from exposure to a particulate-bound nickel concentration of 60 mg/kg acid-extractable nickel at a suspended sediment concentration of 30 mg/L TSS. This study demonstrates that bioavailability of nickel associated with suspended sediment exposure plays a key role in influencing nickel toxicity to corals. These findings assist in assessments of risk posed by increasing nickel mining activities on tropical marine ecosystems.

Highlights

- Field-collected nickel contaminated sediment had no adverse effects on corals or *Symbiodinium* sp.
- Laboratory-spiked nickel sediment and very high concentrations of dissolved nickel caused coral bleaching.
- Significant accumulation of nickel in the coral tissues was observed during the exposure.
- Nickel concentrations in the coral tissues returned to baseline concentrations during the recovery period.
- These findings assist in the management and assessment of risks associated with nickel mining activities in the tropics.

Keywords

Metals; mining; turbidity; bioaccumulation; zooxanthellae; coral bleaching.

1. Introduction

Nickel is an important commodity, predominantly used as an alloying metal in stainless steel production. Global nickel production has experienced near exponential growth since the 1950s, with more than 2 million metric tonnes now being produced annually (USGS, 2017). Economic deposits of nickel occur as either sulfide or laterite ores. With 70% of world nickel reserves being in laterite ores, their extraction is expected to dominate the long-term supply of nickel (Dalvi et al., 2004; Mudd, 2010; Norgate and Jahanshahi, 2011). The majority of laterite ores are found in tropical and subtropical regions (Dalvi et al., 2004). The Southeast Asia and Melanesia region (SEAM), which encompasses three of the world's major nickel producing countries (New Caledonia, Philippines, and Indonesia), is expected to expand its nickel production capacity over the next few decades to meet global demand (USGS, 2017).

Nickel laterite ore is extracted by opencast mining, leading to major landscape alteration, enhanced soil erosion rates, and the production of large volumes of waste material. Tropical rainfall can mobilise and transport large quantities of particulate-bound metals to coastal marine ecosystems as pulsed suspended sediment events. Within the SEAM region, background total suspended sediment (TSS) concentrations in coastal marine waters are typically low (ca. 1 – 5 mg/L) (Browne et al., 2015b; Jouon et al., 2008). However, suspended sediment events can peak at over 100 mg/L TSS in near-shore coastal waters and can sustain moderate concentrations (10 – 50 mg/L TSS) for several days (e.g. Browne et al., 2015a; Hoitink, 2004; Wolanski et al., 2003). Enrichment of coastal sediments with nickel has been documented (Ambatsian et al., 1997; Fernandez et al., 2006; Marchand et al., 2012; Noël et al., 2015), with concentrations as high as 7700 mg Ni/kg reported in surficial coastal sediments (Fernandez et al., 2006) and 2900 µg Ni/L in sediment pore waters (Noël et al., 2015). Elevated concentrations of dissolved nickel have also been reported in near-shore tropical coastal, marine environments, such as 7.1 µg Ni/L reported adjacent to lateritic nickel mining activities in New Caledonia (Fernandez et al., 2017) compared to a background seawater concentration of 0.09 µg Ni/L outside the New Caledonian barrier reef (Moreton et al., 2009).

The SEAM region has unique and highly biodiverse coastal ecosystems with immense ecological and economical significance. It has nearly 100 000 km² of coral reefs in Southeast Asia alone, and the highest level of coral richness and biodiversity in the world (Burke et al., 2002). The projected intensification of lateritic ore mining in the SEAM region has raised concerns as to the potential risk imposed by increased sediment loads and nickel contamination to these vital and fragile tropical marine coastal ecosystems (Gissi et al., 2016).

Limited ecotoxicity data exist for assessing the risk of the adverse impacts of dissolved nickel on coral ecology. Effects data are largely restricted to a few studies that have investigated the effect of nickel on the fertilisation success of scleractinian corals (Gissi et al., 2017; Reichelt-Brushett and Hudspeth, 2016; Reichelt-Brushett and Harrison, 2005). The studies of scleractinian corals have shown that the larval life stage exhibits a broad range of sensitivities to dissolved nickel, but overall hard corals are relatively insensitive to nickel compared to other trace metals such as copper (Gissi et al., 2017). More recent work by Gissi et al. (2019), investigated the effects of dissolved nickel on adult corals and their microbiome, and reported toxicity to the coral only at very high dissolved nickel concentrations (470 µg Ni/L) after a 96-h exposure. Additional data are required to quantify corals' interspecies variability in sensitivity to dissolved nickel. Other work has focused on the physiological response of adult scleractinian coral to nickel enrichment, identifying a beneficial effect

at low nickel concentrations (3.5 µg Ni/L), through stimulating the use of urea as a carbon and nitrogen source, and as a consequence, enhancing both photosynthesis and calcification processes (Biscéré et al., 2018, 2017).

The impact of sediments on corals has been extensively reviewed (Browne et al., 2015a; Erftemeijer et al., 2012; Fabricius, 2005; Jones et al., 2016; Rogers, 1990). Suspended sediment may impact coral by limiting the quantity and quality of light available for photosynthesis in the algal symbionts (*Symbiodinium* sp.) (Anthony and Fabricius, 2000); by sedimentation of the coral surface inhibiting feeding, increasing respiration and leading to a decline in net photosynthetic yield (Junjie et al., 2014; Philipp and Fabricius, 2003); and by impairing recruitment by reducing fertilisation success, larval survival in the water column and larval settlement (Ricardo et al., 2018). Specifically, a decline in photosynthetic productivity of the *Symbiodinium* sp. can lead to coral bleaching (death or loss of the *Symbiodinium* sp. and their associated pigments) (Brown, 1997). Corals have a range of active sediment-clearance mechanisms for dealing with the effects of suspended sediment exposure (e.g. polyp inflation, tentacle action, polyp movement and mucus production) (Stafford-Smith and Ormond, 1992). However, in doing so, they incur an energetic cost, which can lead to sub-lethal effects such as reduced growth, lower calcification rates, reduced productivity, and increased susceptibility to diseases (Erftemeijer et al., 2012). A review of over 30 studies by Browne et al. (2015a) found that tolerance thresholds for coral species to suspended sediment exposure vary widely, over a range of timeframes (hours to one year) and sediment concentrations (10-1000 mg/L TSS).

Few studies have determined the effects of metal-contaminated suspended sediment on corals and none have conducted controlled concentration-response studies and subsequently examined recovery of the coral from environmentally relevant exposures to nickel-contaminated sediments. This study examined the physiological response of the branching coral, *Acropora muricata* (formerly *A. formosa*), following a 7-d continuous exposure to either dissolved nickel; clean suspended sediment; or nickel-contaminated suspended sediment. A laboratory spiked nickel sediment and a field-collected nickel-contaminated sediment were tested to provide different nickel bioavailability scenarios. The response of the coral was determined by quantifying *Symbiodinium* sp. cell densities and loss of colour intensity (i.e. bleaching), as well as measuring nickel concentrations in the coral tissues and algal symbionts. These same endpoints were used to examine the recovery of the coral 7-d post-exposure. This controlled study seeks to determine if there was any additional impact from the nickel associated with the suspended sediments and assists in assessing the potential impacts of the anticipated increased exploitation of lateritic nickel ores in the SEAM region.

2. Materials and Methods

2.1 Coral collection and acclimation

A. muricata is a dominant reef-building species, common to tropical coral reefs throughout the world, including the SEAM region. Large coral fragments were collected from six colonies of *A. muricata* on the 14-16th of June 2017 from Trunk Reef (Queensland, Australia) at 3 to 5 m depth (GBRMPA permit number G12/35236.1). The fragments were further cut into nubbins (4 – 10 cm length) each including one axial corallite, glued (XTRA Loctite super glue, Loctite, Australia) to aragonite plugs and placed in 60-L tanks supplied with flowing seawater (5 L/min) for transport to

the laboratory. The coral nubbins were transferred to 200-L indoor flow-through tanks at the National Sea Simulator (Australian Institute of Marine Science (AIMS), Cape Ferguson, near Townsville, Queensland, Australia) for 7 weeks of acclimation. Acclimation conditions are provided in the supplementary material. Only coral nubbins that were in a healthy condition, i.e. having produced new growth during acclimation, were selected for use in the experiment.

2.2 Sediment collection, preparation and characterisation

To mimic the type of suspended sediment that would be present in a pulse event within the mining impacted SEAM region, fine-grained, organic sediment from two tropical locations in Queensland, Australia were collected for this study. Nickel-contaminated sediment was collected near Saunders Beach (Yabulu, 19°10'14.5"S 146°37'25.4"E), adjacent to a nickel refinery. Uncontaminated sediment was collected from Hartleys Creek, (Wangetti, 16°39'17.3"S 145°34'04.6"E) and was used as a control, as well as for the spiked, high nickel treatments. At both sites, surface sediment (upper 10 cm), was collected using a clean plastic hand trowel and transported back to the laboratory, sieved to remove particles >180 µm, and stored at 4°C. The physico-chemical properties of the sediments before sieving are provided in supplementary Table S1 and S2.

The high nickel treatment was prepared using a two-part nickel-spiking procedure (Hutchins et al., 2008) with the sieved Hartleys Creek sediment. Initially, a super-spike sediment was prepared (10,000 mg Ni/kg nominal) then diluted with clean sediment to achieve the treatment concentration (6000 mg Ni/kg nominal). The treatment concentration was selected to be representative of the high concentrations of nickel that have been reported in coastal sediments near mining activities in the SEAM region (Fernandez et al., 2006; Noël et al., 2015). To prepare the super-spiked sediment, NiCl₂·6H₂O (AR grade, Chem Supply, Australia) was dissolved in filtered (<0.45 µm) clean seawater (Cronulla, 34°04'13.35"S 151°09'25.7"E) and thoroughly mixed into the sediment (200 mL at 171 g Ni/L:3.4 kg sediment dry weight) by hand and allowed to equilibrate at room temperature for 10 weeks, with twice weekly mixing by hand. The super-spiked sediment was then diluted with clean sediment to achieve the treatment concentration and allowed to equilibrate for a further two weeks at 4°C.

Total recoverable metals (TR-M) were determined from a strong aqua-regia digestion by adding a 3:1 ratio of concentrated HNO₃ and HCl (Merck, Tracepur) to a 50-mL Greiner tube containing 0.35 ± 0.05 g (mean ± standard deviation (SD)) of oven dried (110°C for >24 h) sediment. Samples were microwave heated (CEM MARS) at 80°C for 1.5 h. Dilute acid-extractable metals (AE-M) were determined by introducing cold 1 M HCl to 0.5 ± 0.05 g wet sediment in a sediment:acid ratio of 1:50 for 1 h. For both analyses, all samples were measured in duplicate and method blanks and certified reference materials AGAL-10 (Hawkesbury River sediment, Australian Government Analytical Laboratories) were analysed as part of the routine quality control procedures (recovery ≥ 88%).

Sediments were also analysed for total organic carbon (TOC), total inorganic carbon (TIC), particle size and mineralogy both before and after manipulation. Methods for these analyses are detailed in the supplementary material. In this study, particle sizes of <4 µm diameter were considered clay, 4 – 63 µm was silt and 63 – 180 µm was sand.

2.3 Experimental design and conditions

The study consisted of exposing coral nubbins to nine treatments: a water-only control and two dissolved nickel concentrations (0, 200 and 400 µg Ni/L nominal); and six suspended sediment

treatments of clean, nickel-spiked and field-collected nickel-contaminated sediment, each at 5 and 30 mg/L TSS. Exposure duration was 7 d, with coral physiological response variables measured on Days 4 and 7. Post-exposure, there was an additional 7-d recovery period where corals were exposed to non-contaminated seawater and coral response variables were measured again on Day 14. Corals were not fed over the 14 days. The water-only and suspended sediment treatments had two and three replicate tanks per treatment, respectively. One day prior to exposure, 16 – 20 individual coral nubbins were placed on a suspended plastic grating (false bottom floor), 25 cm below the water surface in each replicate tank. First, a unique identification number was randomly assigned to 6 or 9 of the coral nubbins per tank for the water-only and suspended sediment, respectively, by attaching a small piece of paper to the aragonite plug with glue (XTRA Loctite super glue). These select coral nubbins were photographed before being placed in the tank and again when removed (section 2.4.1).

The study was carried out in a controlled environment room at the National Sea Simulator (AIMS), with a sediment dosing and suspension system set-up similar to that of Bessel-Brown et al. (2017). The experiment was carried out in 24 square, clear polyvinylchloride tanks (115 L capacity each) with an inverted pyramid at the base. Each tank was supplied with a constant flow (400 ml/min) of ultra-filtered (<0.04 μm) seawater (temperature 27°C, salinity 35 ppt, pH 8.1) to provide six complete turnovers of water per day. Above each tank was a custom built LED lighting panel, providing even illumination with a mixture of blue and white chips, with a spectral frequency around 460nm. A 12:12 h L:D photoperiod was provided, which consisted of a 2-h period of gradually increasing light in the morning, 8 h of continuous illumination at 100-150 $\mu\text{mol photons/m}^2/\text{s}$, a 2-h period of decreasing light in the afternoon, followed by 12 h of darkness.

The three suspended sediment treatments (clean, nickel-spiked and field-contaminated sediment) were stored in separate 70 L fiberglass tanks and kept in suspension through recirculation at high velocity (3 m/sec) in closed loops around the experimental room by air-operated diaphragm pumps (Sand piper S30). A programmable logic controller (PLC) (Siemens S7-1500) and custom model predictive control logic was used to control the dosing of the tank to the desired TSS concentration. Further details of the experimental sediment dosing and suspension system are provided in the supplementary material.

Concentrated stocks of dissolved nickel (0.5 and 1.0 g Ni/L) were prepared by dissolving $\text{NiCl}_2 \cdot 6\text{H}_2\text{O}$ (AR grade, Chem Supply, Australia) in filtered (<0.04 μm) seawater. Dissolved nickel was then introduced to each of the nickel treatment tanks at a rate of 0.3 mL/min by peristaltic pump to give nominal concentrations of either 200 or 400 $\mu\text{g Ni/L}$. These concentrations were chosen as they occur within the range where measurable effects were observed from dissolved nickel exposures in previous work (Gissi et al., 2019) and to enable comparison with likely nickel concentrations released from the nickel-spiked sediments into the overlying water.

Experimental conditions including water temperature, turbidity and light intensity were continuously monitored, logged and controlled throughout the entire experiment by the PLC. For details see supplementary material. Water quality parameters (including temperature, pH, salinity, specific conductivity and dissolved oxygen) were also measured manually in every tank, every second day (Days 0, 2, 4, 6, 8, 10, 12 and 14) using a YSI Professional Plus meter calibrated according to manufactures specifications (YSI, Yellow Springs, USA).

Sub-samples for dissolved metal analyses were collected from each tank every day during the exposure (Day0 - 7) and every second day during the recovery period (Days 8, 10, 12, and 14) using acid washed (4% v/v HNO₃, Tracepur, Merck) and rinsed (high-purity water) syringes and 0.45 µm polyethersulfone filters (25 mm, Minisart® High Flow PES, Sartorius Stedium Biotech GmbH, Germany). Sub-samples (10 mL) were acidified (0.2% v/v HNO₃) and stored at 4°C.

Sub-samples for dissolved organic carbon (DOC) were collected from 1 replicate tank per treatment on Days 0 and 6. Samples were passed through a 0.45 µm polyethersulfone filter and collected in a glass vial with 2 mL of concentrated H₂SO₄. DOC was measured using a Shimadzu TOC-VCSH/CSN, TOC/TN analyser according to Holland et al. (2018).

2.4 Coral response variables

Coral response variables were measured on Days 4, 7 (at the end of the exposure period) and 14 (at the end of the recovery period). Visual observations of coral behaviour (e.g. polyp activity and mucus production) were also recorded throughout the experiment.

2.4.1 Loss of coral tissue colour intensity (i.e. bleaching)

Coral nubbins were individually removed from the tanks and rinsed with clean filtered (< 0.02 µm) seawater. They were immediately imaged using a high-resolution digital camera (Olympus tough TG4). The coral nubbins were randomly selected from the numbered subset that were imaged pre-exposure so that before and after exposure comparisons could be made. The camera settings and distance of the coral fragment to the camera were standardised throughout the duration of the experiment. The coral images were analysed for colour intensity using the image analysis software ImageJ (Schneider et al., 2012). The method was based on that described by Bessell-Browne et al. (2017) with modification. The outline of the coral nubbins was carefully, digitally traced then the histogram function was used to display the traced area colour intensity in greyscale (black = 0 and white = 255), and the arithmetic mean of the pixel values was recorded. Differences in ambient lighting across images were normalised by analysis of a standard black and white square present in every image collected.

2.4.2 Separating skeleton, tissue and algal symbionts

Following imaging, coral tissue and algal symbionts were removed from the coral skeleton by air-blasting into a clean zip-lock bag (Deschaseaux et al., 2013). The skeleton was placed into a separate zip-lock bag and frozen (for surface area quantification, section 2.4.3). Seawater (5 mL) was added to the bag of tissue and algae, and the slurry homogenised by hand massaging of the bag. This homogenate (2 mL) was then transferred to a 2-mL tube (VWR International, USA) and fixed with 1% v/v glutaraldehyde (25% in H₂O, Sigma-Aldrich, Germany) for quantifying *Symbiodinium* sp. density (section 2.4.3). The remaining homogenate was transferred to a pre-weighed, acid-washed 10-mL centrifuge tube and centrifuged at 1000 g for 10 min to separate the denser algal symbionts from the tissue. The tissue solution was pipetted off the top and transferred to a pre-weighed, acid-washed 50-mL centrifuge tube. Water was removed from the algal symbiont pellet by oven drying at 60°C for ≥24 h and from the tissue solution by freeze-drying (0.28 mbar for 6 h then 0.0010 mbar for ≥ 6 h). Dried samples of coral tissue and algal symbionts were acid digested to determine metal content (section 2.4.4).

2.4.3 Algal symbionts cell density quantification

The cell density of the algae symbionts was determined using a flow cytometer (FACSCalibur, Becton Dickinson, Biosciences) with a 488-nm excitation laser and calibrated with TruCount fluorescent beads (BD TruCount TM tubes, BD Biosciences). An acquisition time of 120 s was set and the flow rate adjusted to achieve > 1000 bead events. Samples were gently homogenised immediately prior to analysis using a tissue grinder and where required, diluted with clean filtered (<0.45 µm) seawater.

Flow cytometry identified two distinct populations of algae isolated from the homogenate. Microscopy revealed that there was likely two genera of algae present, *Symbiodinium* sp. and possibly a Prorocentrum-like dinoflagellate (supplementary Figure S1). Both algal populations were individually quantified by flow cytometry and in the context of cell density measurements will be distinguished hereinafter as *Symbiodinium* sp. and unidentified alga, respectively.

Cell density of the two algal populations was also measured using a haemocytometer for quality control and validation of the flow cytometer values at a frequency of 1 in every 6 samples (n=25). Cell density values obtained by flow-cytometry and haemocytometer were in good agreement ($r^2 = 0.96$ and $r^2 = 0.95$ for *Symbiodinium* sp. and the unidentified alga, respectively) and neither method was found to consistently produce higher or lower densities (supplementary Figure S2). Values obtained from the flow cytometer were converted from cells/mL to cells/cm² using the surface area of the specific coral skeleton from which the algae were extracted.

The surface area of each coral skeleton was determined by 3D scanning. The set-up included an LED projector (ACER), camera (DAVID-CAM-4-M) and powered by the computer software HD 3D Scan 5. A total of sixteen scans (22.5° rotation between scans) were collected for each skeleton with the scanning settings as follows: quality control 0.4-0.6, outlier removal 0.3-1%, close holes 100%, and fusion resolution 500. The visualisation software Autodesk Netfabb was then used to obtain surface area and volume measurements for each coral skeleton.

2.4.4 Nickel accumulation in coral tissue and algal symbionts

The dried algal symbionts and coral tissue samples were pre-digested in a 2:1 mixture of nitric acid (Merck, Tracepur) and hydrogen peroxide (Merck, Emsure) at room temperature for ≥ 18 h. The samples were then microwave heated (CEM MARS) at 60°C for 1 h. After digestion samples were diluted to a nitric acid concentration of 6% (v/v high-purity water). Blanks and certified reference material (0.01 g or 0.10 g to match the sample weights of the algae and tissue, respectively, Dorm-3 Fish protein, National Research Council, Canada) were included with each sample batch to ensure consistency in results achieved (recovery of $89 \pm 33\%$ and $65 \pm 11\%$ for 0.01 g and 0.10 g samples, respectively). Nickel concentrations in the algae symbionts and coral tissue will need to be considered alongside these recoveries; however, the authors note that the algal symbionts and coral tissue were completely dissolved by the digestion process whereas the reference material was not. Baseline concentrations of metals in the coral tissue and algal symbionts were determined from six individual replicate coral nubbins collected prior to the treatment exposures.

The results of the nickel accumulation in the algal symbionts needs to be considered in the context of the presence of both the *Symbiodinium* sp. and the unidentified alga. The authors note, however, that the unidentified alga may have been introduced to the sample from the addition of 5 mL of seawater to produce the homogenate and therefore may not be associated with the coral or may

not have been present during the nickel exposures. Its presence is therefore, if anything, likely to result in a slight underestimation of the nickel concentration in the algal symbionts.

2.5 Metal analyses

The dissolved metal concentrations and digested samples of sediment, coral tissue and algae were measured using inductively coupled plasma-atomic emission spectrometry (ICP-AES, Agilent Technologies 700) or where concentrations were $\leq 5 \mu\text{g/L}$, by inductively coupled plasma mass spectrometry (ICP-MS, Agilent Technologies 7500CE). Multi-element matrix-matched calibration standards (QCS-27, High Purity Standards, USA) were prepared and preserved with 0.2% or 2% v/v HNO_3 (Merck, Tracepur) for dissolved metal samples or digested samples, respectively. A drift standard was run every 10 samples and an in-house independent standard was run with each batch of samples for quality control.

2.6 Statistical analysis

Statistical analyses were carried out in NCSS (NCSS Statistical Software 2007, v07.1.21, USA). The repeated measures ANOVA function was used to assess: 1) if there was an effect of nickel on coral physiological response variables (bleaching and *Symbiodinium* sp. density) by detecting significant differences between the control and each treatment at key time points; and 2) if there was an effect of nickel on bioaccumulation in either the tissue or algal symbionts over time (compared to the baseline measurement) by detecting significant differences within each treatment. The within-group variable was time point (baseline, Days 4, 7 and 14), and the between-group variable was the experimental treatments. Data were tested for sphericity using Mauchly's Test of Sphericity and where the homogeneity of variance assumption was violated, data were adjusted using the Greenhouse Geisser adjustment. Post hoc analysis with the Dunnett's two-sided multiple-comparison test with the control was used to indicate at which time points and in which treatments coral responses were significantly different. An alpha level of 0.05 was used for all statistical tests.

3. Results

3.1 Sediment characterisation

The physico-chemical properties of the sieved suspended sediment treatments are presented in Table 1 and additional chemical properties are provided in supplementary Table S2. The particle size composition of all treatments was relatively uniform and predominately silty (82% 4-63 μm , mean particle size 19-20 μm). TOC concentrations were moderate to high (3.5 – 7.7%). The Hartleys Creek sediment (used as the control) had low concentrations of metals, well below sediment quality guideline values (SQGVs) (ANZG, 2018). The Saunders Beach sediment TR-Ni and TR-Cr concentrations exceeded their respective SQGVs (21 mg/kg and 80 mg/kg) by 11 and 1.3 times, respectively. The AE-Ni and AE-Cr concentrations were 25 and 5% of the TR-M concentrations, respectively, demonstrating that they were mostly present in a mineralised form. Spiking of the sieved (<180 μm) Hartley Creek sediment produced a TR-Ni concentration of 6300 mg/kg and an AE-Ni concentration 6100 mg/kg, close to the target nominal concentration.

Both sediments had a similar mineral composition with the main mineral phases present being silicates (quartz, laboradorite and chlorite), clays (kaolin, illite, mixed layer illite), feldspar group

minerals (albite and microcline), sodium chloride, pyrite, and carbonates (high magnesium calcite and aragonite) (supplementary Figure S3).

Table 1. Characteristics of the tropical sediments used as suspended sediment treatments (sieved to <180 µm), collected from Hartleys Creek (HC) and Saunders Beach (SB), Queensland, Australia.

Sediment	Particle size, µm (%, n=5)			TOC (%, n=1)	Dilute-acid extractable metal (mg/kg, n=3)			Total recoverable metal (mg/kg, n=3)		
	<4	4-63	>63		Ni	Fe	Mn	Ni	Fe	Mn
Clean (HC)	6	82	12	3.5	1	650	7	9	14000	120
Added-Ni (HC)	6	82	12	3.5	6100	540	6	6300	15000	120
Field-Ni (SB)	7	82	11	7.7	60	11000	33	240	23000	100

3.2 Experimental conditions

Water quality was consistent throughout the experiment and across all individual tanks; temperature was $26.8 \pm 0.1^\circ\text{C}$ (mean $\pm$ SD, $n = 192$), pH 8.05 ± 0.05 , salinity 35.9 ± 0.3 PSU, specific conductivity 54.4 ± 0.5 mS/cm and dissolved oxygen 6.3 ± 0.4 mg/L. Dissolved organic carbon ranged from 1.1 to 1.4 mg/L across all treatment tanks (Table S3).

The time-averaged dissolved (<0.45 µm filtered) nickel concentrations in the Dissolved-Ni treatments during the exposure period were equivalent to the target nominal concentrations (Table S3). During the recovery period, dissolved nickel decreased to concentrations close to the detection limit (1.1 µg/L).

Dissolved nickel was below the detection limit in the Clean and the Field-Ni (5 and 30 mg/L TSS) treatments during both the exposure and recovery periods. Dissolved nickel concentrations in the Added-Ni TSS treatments were relatively high (70 ± 30 µg/L and 300 ± 100 µg/L, for the 5 and 30 mg/L TSS treatments, respectively) during the exposure period and were highest towards the end of the exposure period. In Added-Ni TSS treatments the dissolved nickel was much lower (3 ± 3 µg/L and 10 ± 20 µg/L, respectively) during the recovery period, when the corals were exposed to clean seawater for 7 days.

All other dissolved metal concentrations were low across all treatments and time points (As, Cd, Co, Cr, Cu, Fe, Mn, V, Zn were ≤ 1.1 µg/L and Al ≤ 4.4 µg/L, $n=300$, data not shown).

The time-averaged TSS concentrations were close to the nominal values for all treatments: 5 mg/L ($4.3 - 5.1$ mg/L) and 30 mg/L ($27 - 28$ mg/L) (Table S3). Sediment particles were observed to attach to some of the internal surfaces of the tank walls and the false bottom plastic grating, despite the action of the water circulation pumps. No coral nubbins were observed to have considerable sediment deposited on their surface, apart from on those that experienced significant bleaching and polyp retraction and could no longer actively remove the TSS. Algal growth was observed on the plugs of some coral nubbins across all treatments.

Daytime light levels over the 7-d exposure period in the suspended sediment treatments were consistent between treatments, with the time-averaged light level between 120 – 136 $\mu\text{mol}/\text{m}^2/\text{s}$ (Table S3).

3.3 Physiological response of coral to nickel and suspended sediment exposure

All coral nubbins from the control treatments survived the 7-day exposure and 7-day recovery period. Only one fragment across the entire experiment (from the Added-Ni, 30 mg/L TSS treatment) experienced complete tissue necrosis and subsequent sloughing of the tissue from the skeleton. This fragment was not representative of the treatment and so was not included in the analyses.

3.3.1 Loss of coral tissue colour intensity (i.e. bleaching)

Significant bleaching of coral tissue was observed in the Dissolved-Ni treatments and Added-Ni TSS treatments, but not in the Field-Ni TSS treatments (Figure 1 and 2A). A small loss of tissue colour intensity (<10%) was observed in the Control on Day 14, however, this change was not significant and visual inspection indicated that coral nubbins in the control remained in a healthy condition throughout the experiment (Figure 1).

Significant effects of coral bleaching in the Dissolved-Ni treatments were first observed in the 200 μg Ni/L treatment on Day 7 and in both treatments on Day 14. Significantly reduced colour was observed in the Added-Ni 5 mg/L TSS treatment on Day 7 and 14 and in Added-Ni 30 mg/L TSS treatment on Day 7.

3.3.2 *Symbiodinium* sp. density

Baseline measurement of *Symbiodinium* sp. density showed a large range ($5 - 27 \times 10^4$ cell/ cm^2 , $n=6$) for visibly healthy corals. There was no consistent or statistically significant effect on *Symbiodinium* sp. density over the exposure or recovery period in the Dissolved-Ni treatments, Clean TSS or the Added-Ni TSS treatments (Figure 2B). Only in the Field-Ni TSS treatments was there a trend of increasing *Symbiodinium* sp. density over time observed, with the highest cell density observed in the Field-Ni 30 mg/L TSS treatment after 14 days being significantly greater than the Control (Figure 2B)

The presence of the unidentified alga's was highly variable and showed no consistent pattern (Supplementary Figure S4). There was also no relationship between the presence of *Symbiodinium* sp. and the unidentified alga ($R^2 = 0.0054$) (Supplementary Figure S5).

Figure 1. Representative images of coral nubbins taken from each treatment at different times. Each pair of images within a treatment is the same coral fragment imaged prior to exposure (*left image*) and on Day 14 (*right image*) following a 7-d treatment and a 7-d recovery. TSS = total suspended sediment, nominal values.

Figure 2. The influence of treatment condition on bleaching (A) and *Symbiodinium* sp. density (B), before exposure (Baseline), during the exposure (Days 4 and 7) and following a 7-d recovery period (Day 14): *Left* = Dissolved-Ni (0, 200 and 400 µg Ni/L); *Middle* = suspended sediment without Ni (Clean) and spiked with nickel (Added-Ni, 6300 mg total Ni/kg); and *Right* = field-collected nickel-contaminated suspended sediment (Field-Ni, 240 mg total Ni/kg). Values presented are the mean $\pm$ standard deviation (*vertical error bars*), $n=6$ (baseline), $n=2$ (water-only exposure) or $n=3$ (suspended sediment exposures). Significant ($p<0.05$) differences from the control at the corresponding time point are indicated with an asterisk. TSS = total suspended sediment, nominal values.

3.4 Nickel bioaccumulation in coral tissue and algal symbionts

3.4.1 Coral tissue

Significant accumulation of nickel in the tissue of *A. muricata* was observed in the Dissolved-Ni treatments and the Added-Ni 30 mg/L TSS treatment (Figure 3). The tissue of *A. muricata* had significantly accumulated nickel from the Dissolved-Ni treatment on Day 4 in the 200 µg Ni/L and Days 4 and 7 of the 400 µg Ni/L treatment. In both these treatments, the nickel concentrations were highest on Day 4 (2 ± 1 µg/g dry weight (DW) and 3 ± 1 µg/L DW, respectively) then decreased at Day 14. For both Dissolved-Ni treatments, the nickel concentrations had returned to baseline concentrations (0.3 ± 0.2 µg/g DW, $n=6$) following the recovery period. An increase in nickel concentrations in the Added-Ni TSS treatments was observed, but this was only significantly greater than baseline for the Added-Ni 30 mg/L TSS treatment at Day 4 and 7. No nickel was accumulated above the baseline concentration in the tissue of *A. muricata* exposed to the Control, Clean TSS or the Field-Ni TSS treatments.

No changes in the concentrations of other metals (Al, As, Cd, Co, Cr, Cu, Fe, Mn, V, and Zn) compared to the baseline were observed over time (data not shown).

Figure 3. Nickel concentrations ($\mu\text{g/g}$ dry weight) in coral tissue of *Acropora muricata* exposed to: *Left* = Dissolved-Ni (0, 200 and 400 $\mu\text{g Ni/L}$); *Middle* = suspended sediment without Ni (Clean) and spiked with nickel (Added-Ni, 6300 mg total Ni/kg); and *Right* = field-collected nickel-contaminated suspended sediment (Field-Ni, 240 mg total Ni/kg). Values presented are the mean $\pm$ standard deviation (vertical error bars), $n=2$ (Error! Reference source not found.A) or $n=3$ (Error! Reference source not found.B and C). Significant difference ($P<0.05$) from the baseline is indicated with an asterisk. TSS= total suspended sediment, nominal values.

3.4.2 Algal symbionts

The algal symbionts accumulated nickel from the water-only Dissolved-Ni treatments to a maximum of $17 \pm 5 \mu\text{g Ni/g DW}$ (Figure 4), which was 6.8 times greater than the maximum concentration of nickel accumulated by coral tissue in the same treatments (Figure 3). The algal symbionts accumulated nickel from the Dissolved-Ni treatment as measured on Days 4 and 7 in the 200 $\mu\text{g Ni/L}$ and Day 4 in the 400 $\mu\text{g Ni/L}$ treatment. There was large variability in the nickel accumulation in the algae in the 400 $\mu\text{g Ni/L}$ treatment after 7 days exposure. Following the recovery period, the concentration of nickel in the algae in the Dissolved-Ni treatments had decreased and was not significantly different to the baseline concentration ($2 \pm 1 \mu\text{g/g DW}$) pre-exposure.

The nickel concentrations measured in the algal samples from the suspended sediment treatments were confounded by the presence of sediment particles, which also concentrated in the algal cell pellet (supplementary Figure S6). Thus our study was unable to confirm whether or not nickel can be transferred from the sediment and bioaccumulated by the coral-associated algae.

Figure 4. Nickel concentrations ($\mu\text{g/g}$ dry weight) in the algal symbionts of *Acropora muricata* exposed to Dissolved-Ni (0, 200 and 400 $\mu\text{g Ni/L}$). Values presented are the mean $\pm$ standard deviation (vertical error bars), $n=2$. Significant difference ($P<0.05$) from the baseline is indicated with an asterisk.

4. Discussion

4.1 Physiological response of coral to nickel and suspended sediment exposure

Changes in coral tissue colour and *Symbiodinium* sp. density were quantified as a measure of coral stress. Bleaching was observed in response to high Dissolved-Ni exposure (200 $\mu\text{g/L}$ and 400 $\mu\text{g/L}$) in the water-only treatments. These results were comparable to a recent study by Gissi et al. (2019), which observed no effect of dissolved nickel on coral health at 90 $\mu\text{g Ni/L}$ and bleaching at 470 $\mu\text{g Ni/L}$, suggesting that the effects from a short-nickel exposure (4-7 d) occur between 90 – 200 $\mu\text{g Ni/L}$. We caveat this result with the observation that concentrations of nickel greater than 90 $\mu\text{g/L}$ are unlikely to occur in the environment except perhaps for poorly flushed estuarine environments heavily impacted by industrial activities (Angel et al., 2010; de Melo Gurgel et al., 2016; Ezekwe and Edoghotu, 2015; Knauer, 1977).

No bleaching of coral nubbins was observed in *A. muricata* from the Clean TSS treatments. This result is consistent with other studies that have reported tolerance of scleractinian corals to short durations of exposure to TSS ≤ 30 mg/L without significant light limitation or sediment deposition (Browne et al., 2015a; Flores et al., 2012). Bleaching was observed in the directly comparable Added-Ni TSS treatments indicating that a stress response was induced by the presence of the high-concentration nickel-spiked sediment. In the 30 mg/L TSS treatment however, the dissolved nickel (300 ± 100 $\mu\text{g/L}$) was above the concentration where effects were observed for dissolved nickel alone (200 $\mu\text{g/L}$), so it was not possible to determine if the particulate nickel also contributed to the bleaching response. Considering the results of Gissi et al. (2019) who found no bleaching of coral tissue at a dissolved nickel concentration of 90 $\mu\text{g Ni/L}$ following a 4-d exposure, dissolved nickel leached from the sediment (70 $\mu\text{g/L}$) in the 5 mg/L TSS treatment may be below the concentration for effects and the results of the present study provide some evidence that the bleaching effect could be due to a combination of dissolved nickel in the overlying water and particulate-bound nickel on the sediment. Cloran et al. (2010) similarly found reduced survival of the freshwater crustacean *Daphnia magna* exposed to suspensions of nickel-spiked sediment than when exposed to either dissolved nickel or suspended sediment alone. These authors suggested that either dissolved

nickel and suspended sediment acted synergistically to reduce survival or that the test organisms may have also ingested particle-associated nickel (Cloran et al., 2010).

No bleaching of coral tissue was observed in the Field-Ni TSS treatments. Nickel associated with the Field-Ni treatment was highly mineralised (25% acid extractable nickel) and consequently very little nickel was released into solution. Dissolved nickel concentrations in this treatment were close to the natural background concentrations in coastal marine waters ($<1.1 \mu\text{g/L}$ and $1.4 \pm 0.9 \mu\text{g/L}$ for the 5 mg/L and 30 mg/L TSS treatments, respectively) during the exposure period. Therefore, a 7-d continuous exposure of nickel contaminated suspended sediment ($\leq 30 \text{ mg/L}$) at $\leq 60 \mu\text{g AE-Ni/kg}$ ($\leq 240 \mu\text{g TR-Ni/kg}$) was below the concentration to produce effects from nickel for adult *A. muricata*.

During the 7-d recovery period, bleaching continued to occur in the Dissolved-Ni 400 $\mu\text{g/L}$ and Added-Ni 5 mg/L TSS treatments, demonstrating there can be a latent physiological response of the coral following a period of stress. Previous studies have noted that the capacity for corals to recover from bleaching and be repopulated with *Symbiodinium* sp. is dependent on the duration and severity of the stress (Hoegh-Guldberg, 1999). It has been reported that recovery and repopulation of bleached coral tissue by the *Symbiodinium* sp. can take from weeks to several months depending on the extent of tissue damage incurred during the bleaching event (Cunning et al., 2016).

Furthermore, it has been demonstrated that corals exposed periodically to short-duration turbidity events have a greater capacity for recovery from higher levels of sediment exposure and longer durations than corals that infrequently experience suspended sediment events (Nyström et al., 2000) or corals that are constantly exposed to suspended sediment over an extended period of time (such as during dredging) (Browne et al., 2015a).

Changes in coral tissue colour were a more sensitive indicator of coral stress than *Symbiodinium* sp. density. This was most likely due to the naturally high variability in the density of *Symbiodinium* sp. in healthy coral tissue on Day 0. Little to no change in *Symbiodinium* sp. density was observed for the Dissolved-Ni, Clean TSS or Added-Ni TSS treatments. Interestingly, *Symbiodinium* sp. density in the Field-Ni TSS treatment increased over the duration of the experiment (although there was high variability). This result may suggest that the *Symbiodinium* sp. were responding positively to the suspended sediment exposure. The Field-Ni treatments had a relatively high organic carbon content. It has been reported that *Symbiodinium* sp. density and the concentration of photosynthetic pigments may increase in response to enrichment by particulate and dissolved nutrient loads (Tomascik and Sander, 1985). However, such a stimulatory response is not likely to occur at higher TSS concentrations which may cause polyps to retract and reduce their feeding rate (Anthony, 2000).

The mechanism of nickel toxicity to corals and their symbionts is unknown. Limited work on other metals suggests that the symbiotic relationship in adult corals may be disrupted by oxidative stress (Kuzminov et al., 2013). Kuzminov et al. (2013) examined the toxic effect of Cu, Zn, Cd and Pb on *Symbiodinium* spp. and found that the primary targets of metal toxicity were secondary photosynthetic reactions, including electron transport between PSII and PSI and Rubisco activity in the Calvin-Benson cycle.

4.2. Nickel bioaccumulation in coral tissue and coral-associated algae

The pre-exposure, baseline nickel concentration in the coral tissue ($0.3 \pm 0.2 \mu\text{g/g}$ dry weight) was similar to that reported for other scleractinian corals from the Great Barrier Reef (Denton and Burdon-Jones, 1986; Reichelt-Brushett and McOrist, 2003). The pre-exposure, baseline nickel concentration in the algae was an order of magnitude higher ($2 \pm 1 \mu\text{g/g}$ dry weight) than for coral tissue, but lower than previously reported concentrations of $5 - 14 \mu\text{g/g}$ in Reichelt-Brushett and McOrist (2003).

In the Dissolved-Ni treatments, the coral-associated algae had about 7-fold higher nickel content than the coral tissue. This result is consistent with other studies that have reported greater concentrations of metals in the symbionts than the tissues of corals (Ranjbar Jafarabadi et al., 2018; Reichelt-Brushett and McOrist, 2003) and the high metal accumulation efficiency of isolated coral symbionts (Hédouin et al., 2016; Metian et al., 2015). It has been suggested that *Symbiodinium* sp. may play an important role in trace metal regulation by sequestering metals more rapidly than the coral tissue, reducing toxic effects to the invertebrate host (Harland and Nganro, 1990). When under high metal stress the corals may then expel their *Symbiodinium* sp. as a possible mechanism to reduce accumulated metal (Hardefeldt and Reichelt-Brushett, 2015; Harland and Brown, 1989; Harland and Nganro, 1990; Peters et al., 1997).

Following the initial increase in accumulated nickel in coral tissues on Day 4, continuing nickel exposure did not lead to any further uptake on Day 7 (although there was high variability). These results suggest that following an initial uptake of nickel, the corals were able to better regulate the excess nickel in attempting to reach a steady state. Retraction of coral polyps and the production of mucus were both observed responses of the coral in this study to nickel exposure and may have played a role in limiting metal accumulation. Previous studies have identified that retraction of coral polyps in response to metal stress, exposes the coral skeleton which can then directly adsorb metals which are later incorporated into the newly formed skeleton through calcification (Brown et al., 1991). The production of mucus has also been shown to effectively bind heavy metals and thus may be involved in metal regulation (Reichelt-Brushett and McOrist, 2003; Stafford-Smith and Ormond, 1992). In this study, the coral skeletons were not analysed for nickel accumulation due to the short exposure duration and noting that previous work has shown that nickel is only bioaccumulated into newly formed skeleton (Hédouin et al., 2016). Furthermore, the living component of coral (host tissue and algal symbionts) have been shown to display a greater bioaccumulation of metals compared to the skeleton (Hédouin et al., 2016; Metian et al., 2015; Ranjbar Jafarabadi et al., 2018; Reichelt-Brushett and McOrist, 2003).

Following the recovery period, nickel concentrations in coral tissue and the coral-associated algal returned to baseline concentrations by Day 14. These results demonstrate the ability of the corals and algae to depurate nickel during a recovery period. The low retention of nickel in the tissue and algae has been demonstrated in other scleractinian coral species (Hédouin et al., 2016). Using ^{63}Ni , Hédouin et al. (2016) observed a biological half-life of 6.5 days for accumulated nickel following a 14-day exposure.

The amount of nickel accumulated by the coral tissue was greater in the Added-Ni TSS treatment where there was a combined exposure to particulate-bound nickel and dissolved nickel compared to the Field-Ni TSS treatments where exposure was solely from particulate-bound nickel. While the particulate nickel concentration of the nickel-spiked sediment was two orders of magnitude greater

than the field-collected sediment, the most likely explanation for the differences in nickel accumulation observed is the difference in the bioavailability of nickel in the nickel-spiked sediment (97% of TR-Ni was AE-Ni) compared to the field-collected nickel sediment (25% of TR-Ni was AE-Ni) leading to much greater nickel lability and dissolved nickel concentrations in the nickel-spiked sediment. While the greater lability of nickel in the nickel-spiked sediment is most likely due to this sediment being artificially amended with nickel, the greater concentration of iron and TOC in the field-collected nickel-contaminated sediment compared to the nickel-spiked sediment may have also played a role in reducing the bioavailability of nickel in the field-collected nickel contaminated sediment. (Besser et al., 2013; Costello et al., 2016, 2011). A 12-week equilibration period for the spiked sediments was chosen based on previous work, which reported an equilibration period of ≥ 10 weeks is required for nickel partitioning into various solid phases in both marine (Simpson et al., 2004) and freshwater sediments (Brumbaugh et al., 2013). Additionally, the indirect spiking method was used as this has been shown to more rapidly achieve equilibrium of metals in sediments (Hutchins et al., 2008). However, metal equilibration in sediments can be affected by several aspects of the spiking methodology, most notably control of pH and oxidation-reduction potential. The high proportion of total nickel present as AE-Ni in this study demonstrates the need for further research into laboratory nickel spiking procedures. Field-collected contaminated sediments allow for equilibration of metals with the different sediment binding phases over long periods of time, and generally these better reflect metal bioavailability in mine waste over nickel-spiked sediments. This was confirmed in the present study, where no adverse effects on corals or their associated algae were found after exposure to field-collected nickel-contaminated sediments for 7 days.

The amount of nickel accumulated by the coral tissue was greater in the Added-Ni TSS treatments where there was a combined exposure to particulate nickel and dissolved nickel compared to the water-only Dissolved-Ni exposure, so it is likely that both water-borne exposure and TSS uptake via ingestion contributed to the nickel accumulated in the coral tissue. Anthony and Fabricius (2000) found that when light was limited, some corals can switch from being primarily autotrophic to heterotrophic (suspension/detritus). Ingestion of sediment by corals is widely considered to be part of a normal feeding mechanism with corals being capable of assimilating and obtaining nutritional benefits from the associated organic matter (Jones et al., 2016).

4.4 Environmental relevance and implications for risk assessment of nickel in tropical SEAM

As the SEAM region develops additional nickel ore extraction and processing capabilities, specific information is needed to effectively understand the risk posed to important tropical ecosystems such as coral reefs. Reef-building corals, such as *A. muricata*, are important foundation species in tropical environments, providing the structural framework that serves as a habitat for a plethora of other reef-organisms (Peters et al., 1997). Thus, understanding the effects of dissolved and particulate-bound sources of nickel on scleractinian corals is important because changes to their distribution or abundance will have consequences for the stability of entire reef systems and associated importance of these for food safety and food security of local communities. Scleractinian corals and particularly those of the genus *Acropora* have been identified as particularly sensitive to changes in water quality and chemical contaminants making them a suitable first representative tropical test species for use in investigating potential risks from contamination to coral reefs (Peters et al., 1997). It is important to note however, that different coral species and even individuals of the same species that have been exposed to slightly different environmental conditions can have different sensitivities to the same stressor (Anthony and Larcombe, 2000).

Tropical climates are characterised by high seasonal variability, intense storms and high precipitation, which may episodically lead to large volumes of terrestrially derived material or the resuspension of deposited sediment being transported to coral reef ecosystems. In this study the intensity and duration of the suspended sediment exposure was comparable to low (5 mg/L) and moderate (30 mg/L) suspended sediment concentrations that corals could episodically experience in the SEAM region (Browne et al., 2015b; Hoitink, 2004; Jouon et al., 2008; Wolanski et al., 2003). During the experiment, no deposition of suspended sediment on the coral nubbins was observed, apart from on those that experienced significant bleaching and polyp retraction and could no longer actively remove the TSS, and the light levels were the same in all treatments, so the experimental conditions did not confound the effects of the dissolved and particulate nickel. However, in a natural system lack of light and sediment deposition may be co-occurring stressors that may exacerbate the adverse effect of nickel on coral health. For example, Jones et al. (2016) reported that at suspended sediment concentrations comparable to this study (30 mg/L), nearly all light would be attenuated at 5 m depth. Sediment deposition is generally considered more detrimental to corals than turbidity alone because there is an energetic cost for corals to regulate particulate intake, which may lead to down-regulation of photosynthesis, and increased rates of respiration and mucus production (Fabricius, 2005; Philipp and Fabricius, 2003). It is therefore recommended that when using the results of this study, consideration be given to other environmental stressors, both natural and anthropogenic, that may amplify any detrimental effects that may occur and lower coral tolerance to metals.

The mineral composition, particle size and organic content of the sediments used in the experimental suspended sediment treatments were consistent with those reported in suspended sediment measured nearshore from the SEAM region (Fernandez et al., 2017; Ouillon et al., 2010). Previous work has found these finer-grained, organic rich sediments may pose a greater risk to corals than coarser-grained, lighter siliceous and carbonate sediments (Storlazzi et al., 2015; Weber et al., 2006). The nickel concentration of the spiked sediment was chosen to reflect the upper concentrations of nickel reported in sediment in the mining-impacted SEAM region (Ambatsian et al., 1997; Fernandez et al., 2006; Marchand et al., 2012; Noël et al., 2015). Elevated concentrations of dissolved nickel have been reported in near-shore coastal marine ecosystems adjacent to mining or industrial activities (Angel et al., 2010; Fernandez et al., 2017; Knauer, 1977). Our study has demonstrated the importance of considering nickel associated with both particulate and dissolved phases in determining the potential for risk to corals.

At present there are very few nickel sediment toxicity data or whole sediment toxicity tests for tropical marine and estuarine benthic biota (Gissi et al., 2016). This study found that nickel-contaminated sediment may present a risk above that of uncontaminated suspended sediment when the nickel is present in a bioavailable form. For a water-only exposure, the concentration for bleaching to occur was estimated to be between 90 and 200 µg Ni/L considering the results of this study and Gissi et al. (2019). A 7-d exposure of 30 mg/L TSS at 60 mg/kg AE-Ni had no effect on adult *A. muricata*. This information will assist in assessing the potential impacts of the anticipated increased exploitation of lateritic nickel ores in the SEAM region.

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