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Localised and limited impact of a dredging operation on coral cover in the northwestern lagoon of New Caledonia

Mehdi Adjeroud [a,*], Antoine Gilbert [b], Mathilde Facon [b], Marion Foglia [b], Benjamin Moreton [c], Tom Heintz [b]

[a] Institut de Recherche pour le Développement, UMR 9220 ENTROPIE & Laboratoire d'Excellence CORAIL, Université de Perpignan, 52 avenue Paul Alduy, 66860 Perpignan, France
[b] Soproner, 1bis rue Berthelot, BP 3583, 98846 Nouméa Cedex, New Caledonia
[c] AEL Environnement, Anse Vata, 101 Promenade Roger Laroque, 98800 Nouméa, New Caledonia

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ABSTRACT

We report here an interannual survey (2006–2012) of coral cover in the northwestern lagoon of New Caledonia, to assess the impact of an important dredging operation (August 2008–February 2010) associated with the construction of the largest nickel mining site in the Pacific. A BACI (Before-After Control-Impact) analysis failed to detect any significant interaction between period (before, during, and after dredging) and the category of the stations (impact *vs.* control). Among the 31 stations surveyed, only seven showed decreasing coral cover during the study period, mainly due to a decline in Acroporidae. However, the relationship between the dredging and this decrease was highly plausible only for one station, situated 0.9 km from the dredging site. High hydrodynamism in the study area, the abundance of resistant corals and efficient protective measures during the dredging operation might explain these localised and limited impacts.

1. Introduction

Similar to many marine ecosystems, coral reefs have been significantly impacted by chronic or episodic natural and anthropogenic disturbances in recent decades (Hughes et al., 2003, 2007; Bellwood et al., 2004; De'ath et al., 2012). Although some physical and biological disturbances are a routine part of coral reef community dynamics (Grigg and Dollar, 1990), there is concern that the frequency and severity of various types of disturbances have increased over the last three decades (Hoegh-Guldberg, 1999, 2012; Hoegh-Guldberg et al., 2007). Among the local stressors of human origin, dredging operations that are associated with coastal construction, land reclamation, beach nourishment and port construction, represent one of the major sources of disturbance for coral reefs (Brown and Howard, 1985; Salvat, 1987; Rogers, 1990; Erftemeijer et al., 2012).

Dredging can have direct effects through the removal, damage or burial of reef communities and habitats, or *via* indirect lethal or sublethal effects on reef communities through elevated turbidity and sedimentation. These effects can be immediate or progressive and can be observed at various spatial and temporal scales, from localised and temporary effects, to reef-scale and permanent perturbations (Rogers, 1990; Erftemeijer et al., 2012). The resuspension of sediment generally alters light penetration and reduces the growth and calcification of coral colonies (Dodge et al., 1974; Fabricius, 2005) and is also responsible for smothering and abrading colonies (Rogers, 1990). Sedimentation and turbidity stress can also affect settlement and recruitment processes (Fabricius, 2005). It has been demonstrated that the lethal effects of sedimentation are strongly related to the microbial processes that are triggered by the organic matter in the sediment (Weber et al., 2012). Moreover, the resuspension of sediment can release contaminants, thus increasing water pollution and the risk of coral diseases (Haapkylä et al., 2011; Erftemeijer et al., 2012; Burns, 2014; Pollock et al., 2014). Although a large number of scleractinian coral species are sensitive to high sedimentation and turbidity, several species show some adaptation to these stressful conditions (Stafford-Smith and Ormond, 1992; Stafford-Smith, 1993). The extrusion of mesenterial filaments, increased ciliary or polyp activities, tissue expansion and mucus production are some of the major active sediment-rejection mechanisms exhibited by reef corals (Stafford-Smith and Ormond, 1992; Stafford-Smith, 1993; Riegl and Branch, 1995). Thus, the lethal and sub-lethal effects of sedimentation and turbidity associated with dredging can vary both within and among species, and can induce changes at the colony, population and community levels at various spatio-temporal scales (Rogers, 1990). The recovery of coral assemblages following sub-lethal sediment and turbidity stress can take several weeks to months, whereas recovery from lethal stress and mass mortality take years to decades (Pearson, 1981; Erftemeijer et al., 2012).

The New Caledonian barrier reef is the second-longest continuous coral reef in the world after Australia's Great Barrier Reef, and consists

* Corresponding author.
E-mail address: mehdi.adjeroud@ird.fr (M. Adjeroud).

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of a highly diversified coral fauna estimated at 309 zooxanthellate scleractinian species (Pichon, 2007). This high diversity is partly explained by the proximity of New Caledonia to the global hotspot of coral biodiversity, but also by the morphological diversity of its reefs, which includes most reef types (Chevalier, 1973; Andréfouët et al., 2009; Adjeroud et al., 2010; Andréfouët and Wantiez, 2010). Although large-scale natural disturbances are relatively rare compared to other Pacific reefs, such as in French Polynesia or the Great Barrier Reef, New Caledonia is exposed to some localised anthropogenic impacts that are primarily associated with extensive nickel mining that has been present for more than a century (New Caledonia is currently the world's third-largest producer, and nickel exports represent 90 to 95% of the country's exports; Wantiez, 2008; Fichez et al., 2010). Despite this critical situation, few studies on the potential impact of dredging or other types of stress associated with the mining activities of coral communities have been published (Heintz et al., 2015). This lack of understanding prevents effective coastal management and conservation actions that are critically needed, notably since the recent inscription of some New Caledonian reefs on the UNESCO World Heritage list (Wantiez, 2008; Andréfouët and Wantiez, 2010; David et al., 2010).

The aim of the present study was to examine interannual changes (surveyed every November from 2006 to 2012) in scleractinian coral cover among various reef habitats in the northwestern lagoon of New Caledonia, to assess the potential impacts of an important dredging operation. A large navigation channel (4.5 km long, about 7.3 million m^3 of substrate removed; Allenbach et al., 2010) was dug in the lagoon from August 2008 to February 2010, to allow access for large boats to the new industrial harbour. This dredging operation was associated with the construction of the largest nickel mining site in the Pacific (Koniambo Nickel SAS), which began extracting activities in 2014. To reduce the potential effects of this dredging operation, an Environmental Management Plan (EMP), consisting mainly of the use of silt screens around the dredging site and settling ponds and berms in the coastal zone, was set up (Kellog Brown Root Ltd, 2007). The results of the present study might therefore, indicate the efficiency of this EMP.

2. Material and methods

The study area was located in the lagoon of Voh-Koné-Pouembout, in the northwestern part of the main island ('Grande Terre') of New Caledonia (Fig. 1). This area is composed of five distinct reef biotopes (or 'reef types'): fringing reef, reticulated reef, inner barrier reef, outer barrier reef and pass (see a further description of these reef types in Andréfouët et al., 2009). The semi-tropical climate features two seasons generated by the South Pacific Convergence Zone. In summer (November–March), the lagoon water temperature ranges from 26 to 29 °C, whereas in winter (June–August), it decreases to 20 to 22 °C. The lagoon is subject to the influence of both terrestrial and oceanic inputs. Water flow from the ocean derives from either above the barrier reef or through passes (Le Borgne et al., 2010). The dredging zone is close to the Duroc pass, which drives the water exchange. Depending on the tides, currents are alternatively directed inside or outside the lagoon with a mean speed through the pass of $0.4\ m \cdot s^{-1}$ (Kellog Brown Root Ltd, 2007). The lagoon is primarily exposed to south-easterly trade winds that govern the direction of the surface currents, with a predominantly southeast–northwest circulation of surface water in the area (Kellog Brown Root Ltd, 2007). The salinity of the surface water is generally close to ocean salinity, but rainfall can occasionally decrease the local salinity, especially in protected embayments. A negative inshore–offshore turbidity gradient was observed, with mean NTU (Nephelometry Turbidity Unit) measurements being 2.03 ± 0.38, 1.09 ± 0.40, 1.31 ± 1.53, 0.71 ± 0.29, and 0.46 ± 0.16 (mean $\pm$ SD), for fringing reefs, reticulated reefs, inner barrier reefs, pass and outer barrier reefs, respectively (Allenbach et al., 2010). This study area was considered to experience healthy conditions before the beginning of the dredging, as indicated by the diversity and abundance of the fish assemblages surveyed between 2002 and 2007 (Chabanet et al., 2010). For the present survey, a total of 31 study stations were established (Fig. 1). These stations were distributed on the five main reef types, at various distances from the dredging site (Table 1). Considering their distance from the dredging site and the predominantly southeast–northwest surface currents in the area (Kellog Brown Root Ltd, 2007), two categories of stations were distinguished, with 20 stations within the area of potential influence of the dredging operation ('impact' stations) and 11 control ('reference') stations.

Sampling was conducted each year in November, from 2006 to 2012. At each station, the cover of adult coral colonies was recorded along five replicate 20-m transects (Line Intercept Transect Method; see further description in Loya, 1978), parallel to each other. Stations were established between depths of 2 and 9 m (Table 1). Furthermore, we distinguished between Acroporidae and other coral families ('non-Acroporidae'), since Acroporidae are particularly sensitive to sedimentation and turbidity stress. Analyses were conducted at two spatial scales. Firstly, a BACI (Before-After Control-Impact; Stewart-Oaten et al., 1986) analysis was run using the 20 impact and 11 control stations, pooled by reef types (fringing reef, reticulated reef, inner barrier reef, outer barrier reef, and pass). The BACI concept examines the 'before' (pre-dredging baseline) and 'after' (post-dredging) condition of the area, as well as compares 'control' and 'impact' stations over the same period of time. The BACI analysis helps to determine whether changes are due to the dredging itself, or to other factors, such as natural variation in time (see further developments of this method in Osenberg et al., 1994; Underwood, 1994). A two-way analysis of variance (ANOVA) was conducted to test for variation among periods (before, during, after) and categories of stations (impact *vs.* control). Coral cover data (Acroporidae, non-Acroporidae, and total coral) were appropriately transformed (arcsine) to meet the assumptions of normality and homogeneity of variance. At the station level, the interannual variability in these three different categories of coral cover was analysed using non-parametric Friedman tests, as the data did not meet the assumption required for parametric tests, even after appropriate transformations. Post-hoc Wilcoxon tests were performed to determine which pairs of years showed significant differences.

3. Results

The variation in the percentage cover of Acroporidae, non-Acroporidae and total coral before, during, and after the dredging operation at impact and control stations is presented for each reef type (Fig. 2). For fringing reefs, a significant difference was recorded between impact and control stations for non-Acroporidae and total coral cover (Table 2), with higher values in the control zone, whereas the cover of Acroporidae was generally low, with values below 5% at impacted and control stations. Reticulated reefs had a similar total coral cover at impacted and control stations (~12–18%); however, control stations had a higher cover of Acroporidae (~7–12%) than impacted stations (<4%), and consequently, the impacted stations had a higher cover of non-Acroporidae corals (~10–13%). For the inner barrier reefs, significant differences were observed between control and impact stations for Acroporidae and total coral cover, but not for non-Acroporidae. On outer barrier reefs, the percentage cover of Acroporidae and non-Acroporidae corals was different between impact and control stations, and a significant temporal variation was recorded for total coral cover. At the passes, differences between impact and control stations were detected for all categories of coral cover. Despite these differences, no significant interaction between period (before, during, after) and categories of stations (impact *vs.* control) was recorded for any of the reef types and categories of coral cover (Table 2). This indicates that the BACI analysis failed to detect significant effects of the dredging operation when stations were pooled by reef types.

The interannual variability in coral cover was also analysed at each of the 31 stations (see Fig. S1 in Supplementary Information). Three groups of stations can be distinguished: stations characterised by a