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Pat Hutchings, Mireille Peyrot-Clausade, Alicia Osnorno. Influence of land runoff on rates and agents of bioerosion of coral substrates. *Marine Pollution Bulletin*, 2005, 51 (1-4), pp.438-447. 10.1016/j.marpolbul.2004.10.044 . hal-00093841

HAL Id: hal-00093841

<https://hal.science/hal-00093841>

Submitted on 7 Oct 2022

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Influence of land runoff on rates and agents of bioerosion of coral substrates

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Annually large volumes of fresh water laden with sediment are washed down the Daintree River in North Queensland into the Great Barrier Reef lagoon. To investigate the effects of land runoff on bioerosion, samples of recently killed colonies of *Porites* were laid at 6 sites on a cross shelf transect from Snapper Island at the entrance to the river to Osprey Reef, ~328 km from the river mouth out in the Coral Sea. Rates and agents of bioerosion were determined over 4 years and inshore sites exhibited significantly lower rates of total bioerosion than the other sites. Offshore sites experienced high rates of bioerosion primarily due to grazing and internal bioerosion by macroborers such as sponges and bivalves was also important at some of these sites. Inshore sites were covered in heavy layers of silt which inhibited colonization and growth of microborers, primarily algae. This resulted in lower levels of grazing than at offshore sites. However the activity of macroborers (primarily sponges and bivalves) was often high at these sites. The macroboring communities differed between sites and over time and it is hypothesised that these site differences were due to different levels of terrestrial runoff. These results are compared with those from French Polynesia where contaminated terrestrial runoff greatly influenced rates and agents of bioerosion. However other factors such as overfishing may also play an important role in some locations.

Keywords: Bioerosion; Runoff; Sedimentation; Great Barrier Reef

1. Introduction

Coral reefs are dynamic systems representing a balance between reef growth and reef destruction. While numerous studies have focussed on reef growth (Achitov and Dubinsky, 1990) far fewer have investigated reef destruction (MacGeachy and Stearn, 1976; Hutchings, 1986) which includes losses by chemical and physical erosion as well as by biological destruction. These processes can be recognised in both ancient and modern reefs. On a “healthy” reef these processes largely balance one another. Increasingly, as reefs are subject to

anthropogenic impacts, this balance is being destroyed and rates of reef destruction are far exceeding rates of reef growth, resulting in substantial losses of reef framework (Pari et al., 2002; Holmes et al., 2000). Such losses may have significant social and economic consequences, including loss of fish habitats, loss of physical protection from storms and decline in tourism.

Both live and dead coral are subject to biological destruction by grazing and boring organisms, although different suites of organisms are responsible. This study considers only dead coral substrate, which constitutes much reef substrata. Globally, the main agents of grazing are scarids, echinoids and gastropods (Bellwood, 1986; Bellwood and Choat, 1990; Conand et al., 1998; Trudgill, 1983) although on the Great Barrier Reef (GBR) echinoids are conspicuously absent (Clark,

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1938). Borers include both macroborers such as polychaetes, sipunculans, sponges and bivalves and a suite of microborers (Chazottes et al., 1995).

Natural events such as storms often increase the supply of dead corals as do events such as bleaching and plagues of Crown of Thorns starfish. The latter may be facilitated by increased food supplies in the water column resulting from land runoff (see Brodie et al., 2004). Extensive boring of coral colonies may weaken the substrate and make it more susceptible to being dislodged during storm events.

Some studies have suggested that rates of bioerosion of dead coral substrate can be used as a monitoring tool to determine the “health” of a reef (Holmes et al., 2000). However other studies have shown that rates of bioerosion may vary considerably between sites within a reefal system (Kiene and Hutchings, 1994) and within a geographical region (Pari et al., 1998, 2002). Sites regarded as pristine, as well as eutrophic sites may in fact exhibit high rates of bioerosion. All these studies indicate that many factors influence rates and agents of bioerosion: water quality, overfishing, grazing levels, and interactions among these factors. In addition, high levels of microborers in the surface layers of the coral substrate encourages grazing by echinoids (Peyrot-Clausade et al., 2000; Conand et al., 1997, 1998) and scarids (Bellwood, 1996; Bellwood and Choat, 1990; Bruggemann, 1995; Bruggemann et al., 1994a,b). While all these studies have discussed rates of loss by bioerosion, they have by default included unmeasured rates of loss by chemical and physical processes. For example as the substrate becomes bored this increases the surface area available for dissolution by physical and chemical erosion. In this paper, when rates of bioerosion are referred to, they also include losses due to physical and chemical erosion which will vary according to the exposure of the site and probably seasonally.

Over the past decade it has been shown that river flows and associated sediment and nutrient loads are impacting on various components of inshore coastal communities along the Queensland coast, including sea-grass beds (Haynes et al., 2000), and coral communities (Fabricius et al., 2004). The rate of sediment discharge from Queensland rivers has been shown to have increased significantly since European settlement associated with grazing by sheep and cattle (McCulloch et al., 2003) and clearing of native vegetation along the catchment and removal or degradation of riparian vegetation along the rivers (Furnas and Mitchell, 2001). This increasing awareness and concern about the state of health of inshore coastal communities in the GBR lagoon led to the implementation of the Reef Water Quality Protection Plan in 2003 (see Bennett et al., 2004, for details). In addition, recent bleaching episodes have also impacted on inshore reefal communities resulting in an increase in the number of dead coral

colonies (Marshall and Baird, 2000) and with it a potential for an increase in the amount of bioerosion occurring on affected reefs.

In this paper we summarise results from a 1996 to 2000 study to investigate the impact of river runoff on rates and agents of bioerosion. Detailed results after 1 year of exposure can be found in Tribollet et al. (2002) and after 2 and 4 years of exposure in Osnorno et al. (in press). We also include additional results from samples exposed for 1 and 3 years. Finally we compare these findings with those obtained elsewhere on the impacts of terrestrial runoff.

2. Materials and methods

2.1. Study transect

Six sites were selected along a cross shelf transect, 289km in length from the mouth of the Daintree River in far north Queensland coast out into the Coral Sea (Fig. 1). The inshore sites were heavily influenced by river runoff and associated sediment loads from the Daintree River and the other four sites were situated along a gradient of decreasing influence of terrestrial runoff. Two inshore sites were located at Snapper Island (Lat. 16°18'S, Long. 145°30'E) and Low Isles (16°23'S,

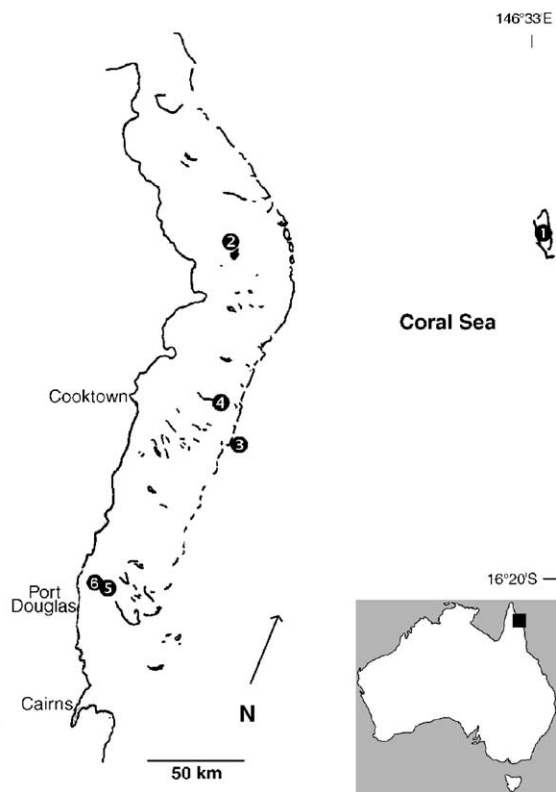


Fig. 1. 1. Osprey; 2. Lizard; 3. Ribbon; 4. Harrier; 5. Low Is; 6. Snapper.

145°34'E); a mid-shelf site was at Lizard Island (14°39'S, 145°27'E); two sites on the outer barrier were at Ribbon Reef N°3 (15°32'S, 145°47'E) and Harrier Reef (15°08'S, 145°42'E) and one site in the Coral Sea 289km from shore was at Osprey Reef (13°54'S, 146°34'E). Detailed information on the sites is given in Tribollet et al. (2002).

The Daintree River was selected as the coastal runoff source because its runoff is primarily sediment rather than sediment and associated fertilisers and pesticides (c.f many other rivers flowing onto the Great Barrier Reef (Neil et al., 2002). Little clearing, no industrial development and limited urbanisation have taken place within the catchment. The Daintree River has a catchment of 2125 km² and Brodie et al. (2003) have modelled the mean annual suspended sediment discharge at 167,000 tonnes based on a measured mean annual volume 943,028 MgI (<http://www.nrm.qld.gov.au/watershed/index>) with marked differences in river flows during wet and dry seasons. During the study period, tropical cyclone Rona in February 1999 (Cheal et al., 2002) produced extensive flooding of the Daintree and greatly increased turbidity of coastal waters (Chin and Ayling, 2002). Both inshore sites along the transect were within the large flood plume produced by this tropical cyclone as well as flood plumes produced annually during the wet season. The inshore site at Snapper Island was located on the well developed fringing reef situated at the mouth of the Daintree River. The inshore site at Low Isles is about 13km south of the Daintree River and also influenced by the flood plumes.

2.2. Experimental design

The experimental substrate consisted of blocks of *Porites*, about 8 × 8 × 5 cm, which had been cut from live colonies collected from reefs on the outer northern GBR. Each block was numbered, measured, soaked in freshwater and the dry weight determined. Blocks showing any sign of borers were discarded. A representative sample of each colony about 5 cm³ was retained and used to determine the density of the coral head from the relationship between dry weight and volume of the sample. Two steel grids of 1 m² were laid at each site in 1996 at a distance of 3–5 m apart, at 7–10 m depth. Each grid was firmly attached to dead coral substrate, and eight blocks were attached equi-distance to each other. Two blocks per grid were collected during each sampling period, after 12, 24, 36 and 48 months in November 1997, 1998, 1999 and 2000, respectively.

2.3. Treatment of samples

Blocks were fixed immediately on collection in 7% formalin and cut in half. One half was dissolved in an acid solution to extract the macroborers, and the other

cut into thin sections 0.7 cm thick for evaluating rates of accretion, grazing and boring. Each slice was photographed and scanned, image analysed, and individual burrows assigned to various groups of borers: molluscs (bivalves and vermetids), sponges and worms (polychaetes and sipunculans) based on their morphology. The amount of substrate removed by each group of bioeroders, expressed as a % of the total amount of internal erosion per slice, was calculated. The amount of accretion by coralline algae per slice was measured and using the density of coralline algae (Laubier, 1962) the rate of accretion in terms of kg m⁻² was measured per block. Rates of loss of calcium carbonate were calculated from the difference between the original dimensions of the slices and the amount remaining after the exposure period. These volumes were converted into rates of loss in kg m⁻² as the density of each coral colony was known. It was assumed that these rates of loss were due to grazing. Net rates of bioerosion were determined from rates of loss by grazers and macroborers and gains by accretion by coralline algae (for more detail see Osorno et al., in press). Rates of removal by microborers were estimated for blocks exposed for 1 year (Tribollet et al., 2002). Macroborers were extracted from the residue remaining after the half block was dissolved, identified to species groups, counted, and measured.

Half of each block collected at Snapper Island after 2 years, which would have been used to determine the density and identify the macrofauna, was lost in the mail. There is, therefore, no data for the density of polychaetes and sipunculans for this sampling period from Snapper Island.

2.4. Statistical analyses

Two-way ANOVA's were used to study the effects of site and duration of exposure on the density of the worms and each group of polychaetes. Estimates were on log transformed ($x + 0.1$) data to homogenize variance found to be skewed (Cochran's test, Sachs, 1984). Multiple comparisons of means using the Student–Newman–Keuls (SNK) test (Zar, 1984) were undertaken to determine which of these means were significantly different from each other. The other groups of non-colonial borers were represented by too few individuals to be analysed.

3. Results and discussion

Significant differences in rates of net bioerosion were found between sites and over time. Rates increased over time and agents of bioerosion differed significantly between sites (Figs. 2 and 3). Inner reef sites had high rates of macroborings and low rates of grazing. In contrast,

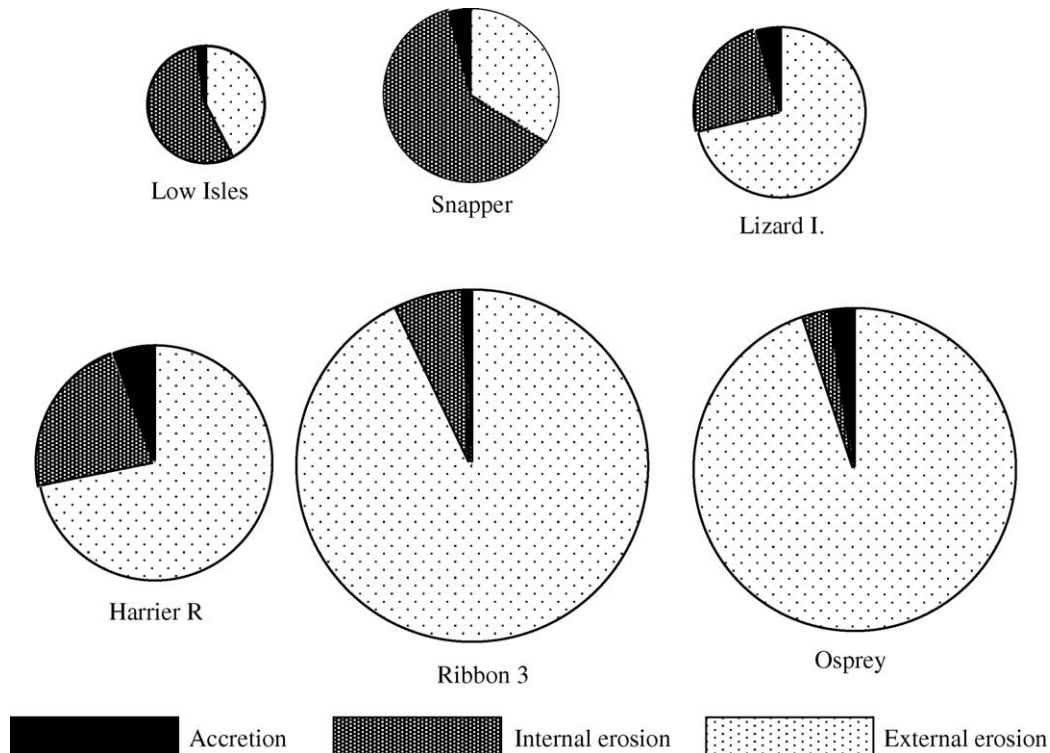


Fig. 2. A representation of the total amount of internal and external erosion and accretion by coralline algae which occurred at each site after 4 years calculated from the scanned slices. (The size of the circle is proportional to the amount of net bioerosion occurring, i.e. net losses were significantly less at Low Isles than at Osprey Reef and the % of grazing increased with distance offshore.)

offshore sites were characterised by high rates of grazing and varying levels of internal erosion by macroborers. Tribollet et al. (2002) found the highest biomass of microborers after 1 year at the offshore sites. This study suggests that these were maintained over time and accounted for high rates of grazing observed at offshore sites. Tribollet et al. (2002) also recorded the lowest levels of microborers at the inshore sites. Again, in this study, we suggest that these patterns were maintained for the following 3 years, and accounts for the low rates of grazing found at these sites. Several studies have shown that species of scarids and echinoids which feed on epilithic and endolithic algae which colonise the surface layers of dead coral substrate select substrates with high biomasses of these algae (Bellwood and Choat, 1990; Bruggemann, 1995; Conand et al., 1998). While no measurements of densities of grazing fish were made during this study, studies by Russ (1984) and Williams and Hatcher (1983) have shown marked differences in these fish populations across the GBR shelf with low biomass of algal grazers in inshore reefs. No grazing urchins were observed during this study and this agrees with other studies which have documented their rarity on the GBR (Clark, 1938; Hutchings, 1986). Elsewhere urchins have been shown to be important grazers of dead coral substrate (Pari et al., 1998; Reaka-Kudla et al., 1996).

Sites at Snapper Island and Low Isles are directly influenced by runoff from the Daintree River. During all sampling periods blocks at these islands were covered in a thick layer of sediment several mm in thickness. This sediment would have inhibited the settlement of both epilithic and endolithic algae. Restricted development of these algal communities would explain the low levels of grazing found at these inshore sites. This in turn may facilitate the recruitment of large numbers of the observed high numbers of macroborers at these sites. Other factors can also be important: high densities of macroborers were also recorded at offshore sites with high rates of grazing. Harrier Reef for example had high rates of internal erosion (sponges, molluscs, polychaetes and sipunculans). Osprey Reef with sponges and molluscs poorly represented had low rates of internal erosion.

These findings imply that the relationship between rates of grazing and abundances of macroborers is not as simple as Kiene and Hutchings (1992, 1994) suggested. They speculated that continual removal of the surface layers by grazers restricts larval settlement and colonisation by macroborers. Other factors may also be important, such as local patterns of recruitment via pelagic larvae (Hutchings, 1986). This varies between windward and leeward sites as well as between years (Hutchings et al., 1992). Here, we have shown that even

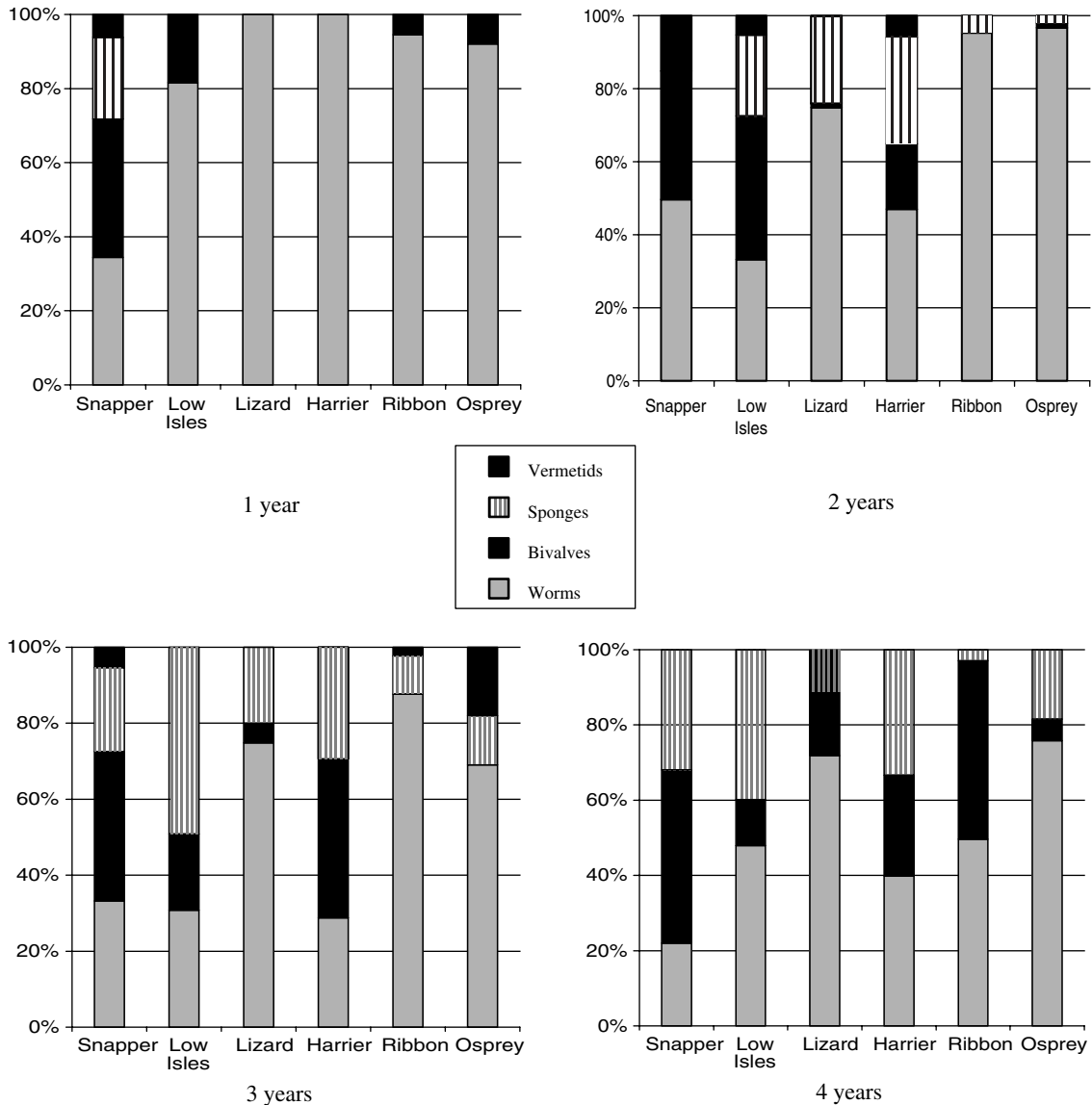


Fig. 3. Average losses attributed to each group of internal borers expressed as the % lost from replicate scanned slices for each site after 1, 2, 3 and 4 years of exposure.

within a site considerable variation occurs between blocks and between grids within a site, indicating that factors operating at a very small scale are important in determining recruitment patterns. The stages of larval settlement and metamorphosis are extremely vulnerable to predation by grazers that remove the surface layers of the coral substrate in search of epilithic and endolithic algae. Gradually, as these macroborers penetrate deeper into the substrate they become less susceptible to such predation.

The contribution of the various groups of borers to bioerosion varied significantly between sites and over time. Sponges were poorly represented at all sites after 1 and 2 years but after 4 years they were responsible for significant bioerosion at the two inshore sites (-0.322 ± 0.477 ; $-0.338 \pm 0.333 \text{ kg m}^{-2}$) and Harrier

Reef ($-0.303 \pm 0.276 \text{ kg m}^{-2}$), and least represented at Ribbon Reef N° 3 ($-0.012 \pm 0.020 \text{ kg m}^{-2}$) and Osprey Reef ($-0.031 \pm 0.038 \text{ kg m}^{-2}$). Similarly, rates of boring by bivalve molluscs were initially low but after 4 years, rates were highest at Snapper Island ($-0.505 \pm 0.544 \text{ kg m}^{-2}$) and Ribbon Reef No 3 ($-0.473 \pm 0.598 \text{ kg m}^{-2}$) and lowest at Osprey Reef ($-0.012 \pm 0.035 \text{ kg m}^{-2}$) and Lizard Island ($-0.088 \pm 0.170 \text{ kg m}^{-2}$). Vermetids (gastropod molluscs) were poorly represented at all sites except at Snapper Island after 2 years.

Boring rates by worms (polychaetes and sipunculans) exhibited no significant differences between sites, but they did increase significantly over time. The composition of the worms did vary between sites (Osnorno et al., in press) with different species of sipunculans

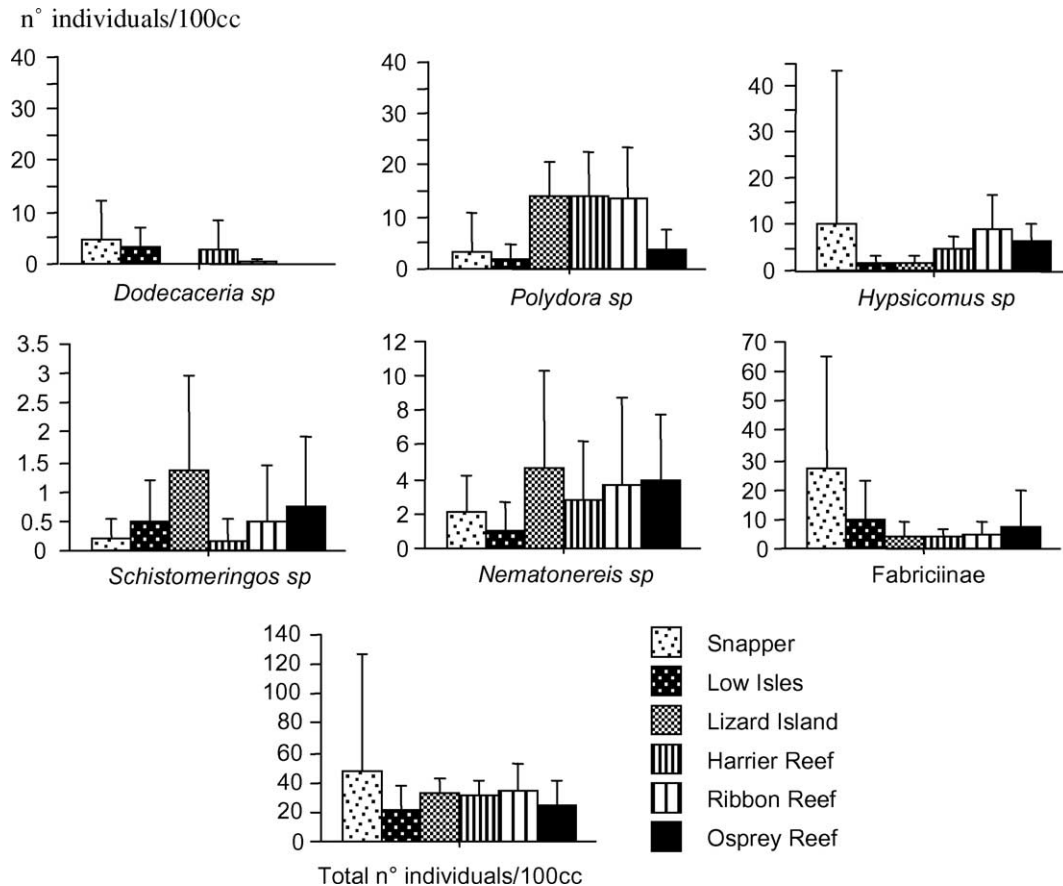


Fig. 4. The distribution of total numbers of individual polychaete species and total numbers of polychaetes present after 4 years at each site expressed as no per 100cm³.

being dominant at particular sites. Significant differences between sites occurred in the total number of polychaetes present after 4 years (Fig. 4). Sites fell into three groups: Low Isles had the lowest number, Snapper Island and Osprey Reef had the most and the other 3 sites had intermediate values (Tables 1 and 2). There were significantly fewer individuals after 1 year than after years 2, 3 or 4 (Table 2). Some individual groups of polychaetes exhibited no significant time or site effects (e.g. *Arabella*, *Lysidice* and *Eunice* spp.) *Polydora* spp. exhibited puzzling site effects. It has low densities at Snapper Island, Low Isles and Osprey Reef, and high densities at the other sites (Table 1). *Polydora* may be either a filter or surface deposit feeder and its avoidance of inshore sites with abundant sediment is understandable. But the reason for its poor representation at the furthest offshore site is not obvious, and may be an artefact of small sample size. *Dodecaceria* was uncommon until after 2 years and became common at Low Isles and Snapper Island and to a lesser extent at Harrier Reef. *Dodecaceria* is a deposit feeder and thick layers of sediment in inshore reefs favour dense populations of this species. The sabellid *Hypsicomus* spp., (a filter feeder), was rare at the 2 inshore sites

and Lizard Island. There was considerable variability at Snapper Island with one block collected after 3 years containing 116 individuals per m⁻³ (c.f. <10 individuals per m⁻³ were usual). This represents one extremely successful recruitment event and highlights the patchy nature of recruitment. The small filter feeding Fabriciinae was abundant at Snapper Island and *Schistomeringos* (probably a carnivore) was commonest at Lizard Island. Similar patterns of polychaete distribution were found at high and low islands in French Polynesia which were subjected to varying levels of terrigenous runoff (Hutchings and Peyrot-Clausade, 2002).

Rates of accretion by coralline algae were relatively low initially but increased over time. Maximum rates occurred at Osprey and Harrier Reefs but surprisingly, rates at clear water offshore Ribbon N° 3 and Lizard Island were similar to those recorded at turbid Snapper Island. The lowest rates were recorded at Low Isles. One would have expected the environmentally similar Snapper Island and Low Isles to be biologically similar and to have consistently lower rates than the other sites which are situated further offshore away from any flood plumes of the Daintree River.

Table 1

Results of 2-way ANOVA's on arcsine square root transformed data of mean % of slices eroded by all macroborers and for each major group of macroborers

	df	MS	F	P	Sites	SN	K		
Site	5	131.08	9.45	0.0001	Ribbon	a		12months	a
Time	3	968.84	69.9	0.0001	Lizard	a		24months	a
Site * Time	15	49.92	3.6	0.0001	Osprey	a		36months	b
Residual	261	13.86			Harrier		b	48months	c
					Low Isles		b		
					Snapper		b		
<i>% total internal bioerosion.</i> Significant differences in the % of total internal erosion occurred between sites. An SNK test revealed 2 groups of sites. Snapper, Low Isles & Harrier have significantly more erosion than other sites. Amount increases significantly over time.									
Site	5	6.8	2.34	0.0420	Low Isles	a		12months	a
Time	1	296.37	102	0.0001	Osprey	a	b	24months	a
Site * Time	15	9.10	3.13	0.0001	Ribbon	a	b	36months	b
Residual	261	2.9			Snapper	a	b	48months	b
					Lizard	a	b		
					Harrier		b		
<i>% of surface eroded by polychaetes and sipunculans.</i> An SNK test revealed 3 groups of sites. Maximum erosion by polychaetes & sipunculans occurs at Harrier and minimum at Low Isles, these % are significantly different to all other sites. Amount increases significantly over time.									
Site	5	113.38	7.05	0.0001	Ribbon	a		12months	a
Time	1	210.73	13.01	0.0001	Osprey	a	b	24months	a
Site * Time	15	39.23	2.44	0.0024	Lizard	a	b	36months	b
Residual	261	2.9			Snapper		b	48months	b
					Harrier		c		
					Low Isles		c		
<i>% of surface eroded by sponges.</i> An SNK test revealed 4 groups of sites. Maximum erosion by sponges occurred at Harrier and Low Isles, and minimum at Ribbon. Amount increases significantly over time.									
Site	5	116.08	11.5	0.0001	Osprey	a		12months	a
Time	1	245.3	24.2	0.0001	Lizard	a		24months	a
Site * Time	15	39.9	3.94	0.0001	Ribbon		b	36months	b
Residual	261	10.11			Low Isles		b	48months	c
					Harrier		b		c
					Snapper		c		
<i>% of surface eroded by molluscs.</i> An SNK test revealed 4 groups of sites. Maximum erosion by molluscs occurred at Snapper and least at Osprey & Lizard, these are significantly different to all other sites. Amount increases significantly over time.									

This study shows that after 4 years, there was up to four times the erosion at offshore reefs than was observed at nearshore reefs. The highest rates of net bioerosion (accretion minus erosion due to both grazing and boring) were found at the offshore sites well away from terrestrial influence. The highest rates of loss were found at Ribbon Reef N° 3 ($-7.846 \pm 3.218 \text{ kg m}^{-2}$) followed by Osprey Reef ($-4.918 \pm 2.076 \text{ kg m}^{-2}$) and Harrier Reef ($-3.397 \pm 2.261 \text{ kg m}^{-2}$) and the lowest at Low Isles ($-1.090 \pm 0.499 \text{ kg m}^{-2}$) and Snapper Island ($-1.590 \pm 0.773 \text{ kg m}^{-2}$) and Lizard Island with intermediate values ($-1.459 \pm 0.679 \text{ kg m}^{-2}$) (Fig. 1). The components of bioerosion vary between environments and even among sites with similar environments (e.g. the outer barrier Ribbon N° 3 and Harrier Reef) or inshore sites (e.g. Low Isles and Snapper Island). These major differences in rates of net bioerosion are due mainly to high rates of grazing at the three sites furthest offshore which are not subjected to flood plumes from the Daintree River (Fig. 1). This contrasts with the findings of Pari et al. (1999) Pari et al., 2002) who found extremely high rates of external erosion by grazing ($-6.87 \pm 2.16 \text{ kg m}^{-2}$)

at an inshore site at Faaa adjacent to Papeete in Tahiti, French Polynesia. A small stream flows into the lagoon close to this site and is contaminated with industrial and domestic waste (Fraizier et al., 1985) plus some sediment associated with clearing in the catchment. Extensive overfishing has occurred at this site and dense populations of echinoids were present, the development of which is encouraged by the lack of fish. Another site of Pari et al. (2002) at Atimaono subjected to heavy sedimentation but without excessive nutrients had low rates of external erosion ($-1.58 \pm 0.27 \text{ kg m}^{-2}$) due to low densities of echinoids. We suggest that the sediment at Atimaono restricted the development of endolithic and epilithic algal communities. This suggests that the presence of nutrients allows dense populations of epilithic and endolithic algae to develop and this in turn supports dense populations of echinoids and associated high rates of grazing (Pari et al., 2002). Le Bris et al. (1998) measured water quality around Tahiti and found that levels of organic matter were much higher at Faaa than at Atimaona. At no time during the 5-year study were blocks from this site in Tahiti covered with a thick layer of sed-

Table 2
Results of 2-way ANOVA's on log transformed total no of polychaetes and major polychaete borers

	df	MS	F	p	Sites	SN	K	Time	S	N	K
Log total individuals/100 cm ³											
Site	5	0.1766	2.925	0.0187	Low Isles	a		1 year	a		
Time	3	0.4408	7.303	0.0003	Osprey	a	b	4 years		b	
Site * Time	14	0.1602	2.655	0.0038	Snapper	a	b	2 years		b	
Residual	69	0.060			Harrier		b	3 years		b	
					Ribbon		b				
					Lizard		b				
Log <i>Dodecaceria</i> sp.											
Site	5	0.7519	13.120	0.0001	Lizard	a		1 year	a		
Time	3	0.2282	3.982	0.0112	Osprey	a		2 years		b	
Site * Time	14	0.2320	4.048	0.0001	Ribbon	a		3 years		b	
Residual	69	0.0573			Harrier		b	4 years		b	
					Low Isles		b				c
					Snapper		b				c
Log <i>Polydora</i> sp.											
Site	5	2.091	17.971	0.0001	Snapper	a					
Time	3	0.093	0.801	0.497	Low Isles	a					
Site * Time	14	0.158	1.357	0.198	Osprey	a					
Residual	69	0.116			Ribbon		b				
					Harrier		b				
					Lizard		b				
Log <i>Fabriciinae</i>											
Site	5	0.7000	3.552	0.0065	Lizard	a					
Time	3	0.1480	0.751	0.5252	Harrier	a					
Site * Time	14	0.2419	1.227	0.2763	Osprey	a					
Residual	69	0.1970			Ribbon	a					
					Low Isles	a					
					Snapper		b				
Log <i>Schistomeringos</i> sp.											
Site	5	0.1255	3.136	0.0131	Harrier	a					
Time	3	0.0791	1.978	0.1252	Snapper	a					
Site * Time	14	0.0330	0.825	0.6397	Ribbon	a	b				
Residual	69	0.0400			Low Isles	a	b				
					Osprey	a	b				
					Lizard		b				
Log <i>Nematonereis</i> sp.											
Site	5	0.1924	1.556	0.184				3 years	a		
Time	3	0.6415	5.1895	0.0027				2 years		b	
Site * Time	14	0.2720	2.200	0.0162				1 year		b	
Residual	69	0.1236						4 years		b	
Log <i>Hypsicomus</i> sp.											
Site	5	0.9728	8.763	0.0001	Snapper	a					
Time	3	0.1900	1.712	0.1726	Low Isles	a					
Site * Time	14	0.0789	0.711	0.7555	Lizard	a					
Residual	69	0.1110			Harrier		b				
					Osprey		b				
					Ribbon		b				

iment as found at Snapper Island and Low Isles. It would be extremely interesting to repeat the experiments on the GBR but at the mouth of another river which has both high sediment and nutrient loads, which lacks dense populations of echinoids and reasonable populations of grazing fish still occur. It is not obvious what the findings would be.

In interpreting results from bioerosion studies it is essential to distinguish between natural fluctuations in

rates and agents of bioerosion which have been demonstrated to occur in this study and others (Kiene and Hutchings, 1992, 1994) and fluctuations due to anthropogenic influences (Pari et al., 2002). Typically there are several anthropogenic inputs acting together. At Faaa, elevated levels of silt and nutrient runoff as a result of land clearing and in addition fish populations have been severely depleted which has allowed large populations of echinoids to become established. All

these factors also inhibit coral recruitment and on this reef, rates of bioerosion far exceed those of reef growth and substantial losses to the reef framework is occurring (pers observation).

So by considering the results from the cross shelf transect on the GBR with other studies one can propose three hypothesis of the impact of land runoff on bioerosion. First, sediment-laden water, which is not contaminated by high levels of fertilisers and nutrients protects coral substrates from heavy grazing while still allowing macroborers to recruit (this study). Second, in contrast sediment plus nutrients encourages algae, macro and microborers, as well as facilitating high rates of grazing (Pari et al., 2002). Third, nutrients alone do not necessarily produce any significant impact on levels of bioerosion (Kiene, 1997; Koop et al., 2001). In this latter study which was part of the ENCORE study (Koop et al., 2001) elevated nutrient loads may have been too low and/or taken up too fast or dispersed too quickly to affect bioeroder abundance. Additional studies by Osorno (in prep.) on slices of *Porites* exposed during the same ENCORE project have, however, found significant impacts of nutrients on rates and agents of bioerosion.

In conclusion, the concept of using rates of bioerosion as indicators of the reef health as suggested by Holmes et al. (2000) is laudable. However measures do not have a simple relationship to runoff effects. We clearly need to know more about the naturally occurring fluctuations as well as anthropogenic impacts on rates and agents of bioerosion. At this stage it appears that it is rates of grazing that are most susceptible to change associated with anthropogenic impacts not internal rates of boring by macrofauna.

Acknowledgments

We would like to acknowledge the logistic support provided by “Undersea Explorer” and especially Andy Dunstan and John Rumney. Kate Attwood of the Australian Museum helped in the preparation of the samples, in sorting some of the samples and figure preparation.

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