Short and intermediate-term consequences of environmental disturbances on coral reef fish abundance (#27764)

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Short and intermediate-term consequences of environmental disturbances on coral reef fish abundance

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Global warming induces an increase in frequency and intensity of extreme weather events such as cyclones, storms and prolonged heatwaves. For instance, the Great Barrier Reef around Lizard Island has suffered from severe tropical cyclones Ita (2014) and Nathan (2015) and a massive coral bleaching due to El Niño (2016). Here, we asked how fish communities evolved in response at two reef sites, both affected by bleaching but only one harbouring a heavily damaged reef structure since the cyclones. We quantified fish abundance and classified the species according to their functional trait (i.e., diet composition). We compared baseline data from before the disturbances to data collected after the disturbances in 2016 and 2017. Overall, we recorded up to 78% declines in fish densities after the environmental perturbations. The decrease in densities was more substantial in 2017 than in 2016, in particular at the site affected only by the 2016 bleaching. At the site damaged by cyclones and bleaching, the overall decline was due to significant reductions in fish densities in nine of eleven fish functional groups. Furthermore, at the site affected by bleaching and not by cyclones, we recorded two functional groups that showed significant declines in 2017, as well as increased piscivores densities. Altogether, environmental perturbations due to extreme climate events appear to have detrimental consequences for reef fish populations that may accumulate over several years.

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Abstract

Global warming induces an increase in frequency and intensity of extreme weather events such as cyclones, storms and prolonged heatwaves. For instance, the Great Barrier Reef around Lizard Island has suffered from severe tropical cyclones Ita (2014) and Nathan (2015) and a massive coral bleaching due to El Niño (2016). Here, we asked how fish communities evolved in response at two reef sites, both affected by bleaching but only one harbouring a heavily damaged reef structure since the cyclones. We quantified fish abundance and classified the species according to their functional trait (i.e., diet composition). We compared baseline data from before the disturbances to data collected after the disturbances in 2016 and 2017. Overall, we recorded up to 78% declines in fish densities after the environmental perturbations. The decrease in densities was more substantial in 2017 than in 2016, in particular at the site affected only by the 2016 bleaching. At the site damaged by cyclones and bleaching, the overall decline was due to significant reductions in fish densities in nine of eleven fish functional groups. Furthermore, at the site affected by bleaching and not by cyclones, we recorded two functional groups that showed significant declines in 2017, as well as increased piscivores densities. Altogether, environmental perturbations due to extreme climate events appear to have detrimental consequences for reef fish populations that may accumulate over several years.

Introduction:

The recently observed increase in frequency and magnitude of extreme weather events are attributed to the anthropogenic global warming (Cai et al., 2014; Pachauri et al., 2016; Cheal et al., 2017; Hughes et al., 2018). A naturally occurring climate event is the El Niño cycle; it brings

warm water towards the indo-pacific provoking hence a cascade of changes in the weather conditions. A prolonged El Niño event would lead to a remarkable increase in seawater temperatures (Cai et al., 2014; Hoegh-Guldberg & Ridgway, 2016). Overstressed coral tissues, would, therefore, expulse their intracellular symbionts "zooxanthella", from where corals gain their different pigmentations. Henceforth, bleached corals face death fate if they do not re-establish the symbiotic relationship with the zooxanthella within a range of six months post-bleaching (DiazPulido & McCook, 2002). In addition to the threat of coral bleaching, cyclones can be extremely devastating due to the formation of strong waves that can ravage exposed coral reef fields, including organisms living there (Cheal et al., 2017). The recovery might be compromised if the reef is repeatedly exposed to tropical cyclones over short-time intervals (De′ath et al., 2012; Puotinen et al., 2016).

Lizard Island, an island in the northern Great Barrier Reef (GBR), Australia, is within a marine reserve, a world-top destination for marine biologists. In April 2014, Cyclone Ita hit Lizard Island (Pizarro et al., 2017), reaching an intensity of the highest level in tropical cyclones categories (Puotinen et al., 2016). In April 2015, Lizard Island got hit by another severe tropical cyclone, Cyclone Nathan (Pizarro et al., 2017). Furthermore, the Australian 2016 summer (i.e., December to February) witnessed the worst massive coral bleaching event since the 1980's (Hughes et al., 2017), due to the warmest El Niño event recorded ever (Normile, 2016). The bleaching touched more than 60 % of the coral cover at the GBR (Hughes et al., 2017). Thus, for the third year in a row, Lizard Island suffered from extreme weather events, raising concerns about potential consequences.

Here we provide data on fish abundance at two different time points (i.e., June 2016 and June 2017) after the perturbations at Lizard Island on two reef sites for which we had data before the perturbations. We compared fish total abundance as well as fish species categorised to functional group level before and after the perturbations. To sort fish species into functional groups, we opted for diet as the functional trait. Here, we expected to find a significant decline in fish species that rely either directly or indirectly on live corals for their diet (Wilson et al., 2006). In contrast, due to the colonisation of dead corals by microalgae (Cheal et al., 2010), we expected an increase in the abundance of herbivorous fish species specialised to feed on such algae (Randall, 1961). Finally, there was an important difference between our two study sites in that the cyclones destroyed the reef structure only at Site-1 while Site-2 had been sheltered from wave actions by the island itself. The destruction of reef structure implied the destruction of shelters, leading us to predict that predators should also belong to winners of the perturbations at Site-1 by gaining easy access to prey.

Methods:

Field sites:

Fish census data were collected on reefs around the Lizard Island Research Station (14.6682° S, 145.4604° E), Great Barrier Reef, Australia. The two study sites are Mermaid Cove as "Site-1" and North Horseshoe as "Site-2". Site-1 forms a continuous fringing reef approximatively of a size of 35,000 m^2 (depth 1 to 7 m) (i.e., estimated from maps: https://www.freemaptools.com/area-calculator.htm), located in a small bay on the northern side

of Lizard Island, this part of the island is an exposed area (Ceccarelli, Emslie & Richards, 2016; Pizarro et al., 2017). Site-2 is also a continuous reef of a coral garden (depth 1 to 4 m) of a size approximately 17,000 m^2 located within a protected area on the western side of the island (see Fig. 1) (Pizarro et al., 2017).

Cyclone Ita and Nathan severely damaged the exposed Site-1, and the remained coral cover got bleached through the coral bleaching event. However, Site-2, due to its protected location from the two cyclones was touched by the bleaching only, in 2016. Data before the extreme weather events were collected between June and August 2011 from Site-1 (Wismer et al., 2014) and between June and August 2014 from Site-2 (Triki et al., 2017), whereas data after the extreme weather events were collected between June and August in 2016 and 2017 from both sites Site-1 and Site-2 (Fig. 1).

Data collection followed a timeline concerning the extreme weather events: In 2011: fish census data were collected by Wismer et al. (2014) at Site-1. In May 2014: Cyclone Ita hit Lizard Island, only Site-1 got damaged by the cyclone. In June 2014: fish census data were collected Site-2 (Triki et al., 2017). In May 2015: Cyclone Nathan hit Lizard Island, and again only Site-1 got hit. In February/March 2016: El Niño event bring warm water causing coral bleaching, here both sites are touched by the bleaching. In June 2016: we collected fish census data from both sites. In June 2017: we went back to both sites and collected fish census data. All fish census data collection were conducted in the Australian winter, excluding variation in seasons as a confounding variable.

105 Fish census data collection:

106

107 Scuba divers collected fish censuses data to estimate coral reef fish abundance. Therefore, we
108 swam (n=10) replicate of 30m transects at each site/year. We placed the transects haphazardly
109 either parallel the shoreline or to the reef crest. We first recorded all large visible fish with a
110 body total length TL > 10cm on a 5m wide area, followed by small fish with a body TL ≤ 10cm
111 on a 1m wide area along the 30 m transect. We recorded only adult coral reef fish. Each of the
112 ten replicate transects, on each site/period, were sampled at least 10 meters apart from each
113 other. Overall, we covered an area of 1500 m^2 on each site/period. All fish were identified to
114 species level, and census protocols followed (Wismer et al., 2014; Triki et al., 2017). We
115 calculated the fish abundance by scaling densities per 100 m^2.

116

117 Fish species categorisation:

118

119 Fish species were categorised into functional groups based on diet (i.e., all species that share
120 similar trait value (Butterfield & Suding, 2013; Brandl et al., 2016). Overall, we had 11 groups
121 (Frédérich et al., 2009; MacNeil et al., 2015; Froese & Pauly, 2016; Brandl et al., 2016): (1)
122 browsers, fish that mainly feed on macroalgae, for example: bluespine unicornfish, *Naso*
123 *unicornis*; (2) corallivores, fish that feed on corals, for example: golden butterflyfish, *Chaetodon*
124 *aureofasciatus*; (3) detritivores, fish that mainly feed on dead organic material "detritus" , for
125 example: striated surgeonfish, *Ctenochaetus striatus*; (4) scrapers/excavators, fish that would
126 remove reef substrate while looking for living material, for example: daisy parrotfish, *Scarus*
127 *sordidus*; (5) grazers, fish that feed on the fast growing turf algae, for example: barred

rabbitfish, *Siganus doliatus*; (6) macro-invertivores, predators that feed on large invertebrates, for example: orange-lined triggerfish, *Balistapus undulatus*; (7) micro-invertivores, fish that feed on small invertebrates, for example: batu coris, *Coris batuensis*; (8) pisci-invertivores, predators that feed on fish and invertebrates, for example: longface emperor, *Lethrinus olivaceus*; (9) piscivores, predators that feed on fish, for example: Honeycomb grouper, *Ephinephelus merra*; (10) planktivores, fish that feed on plankton, for example: scissortail sergeant, *Abudefduf sexfasciatus*; (11) spongivores, fish that feed on the sea sponges, for example: sixbar angelfish, *Pomacanthus sexstriatus*. Categorisation into functional groups followed methods of Wernberg et al. (2013) and MacNeil et al. (2015), respectively. Furthermore, we completed missed information for fish species that do not figure in the two studies with data from a web-based FishBase (Froese & Pauly, 2016).

Statistical analyses:

All data analyses and figures were generated by using the Software Rstudio® (version darwin.10.08.0). Due to data violating assumptions regarding normality and homogeneity of variances, we opted for non-parametric statistics. We considered data collected from the same site but in different years as independent samples due to the time lapse between the three periods of data collection. We ran Kruskal-Wallis from the package (*agricolae*) in R language where we tested for changes in fish densities in each affiliation group in each year of data collection within each study site. To correct for multiple hypotheses tests on data from each site separately, we opted for the sequential Holm-Bonferroni method to adjust the threshold of probability significance α. Significant Kruskal-Wallis *p*-values were tested for post hoc analyses to detect

151 eventual differences occurred between the time periods within each site. Also, for the post hoc
152 analyses, we corrected for multiple tests between years by employing Holm's method in the
153 *agricolae* package in R language.
154
155 **Ethical note**
156 The Animal Ethics Committee of the Queensland government (DAFF) approved the project (CA
157 2016/05/970 and CA 2017/05/1063).
158
159 **Data availability:**
160 The data is available in the repository figshare (Data DOI: 10.6084/m9.figshare.4990919).
161
162 **Results:**
163
164 Fish census data showed that fish abundance significantly changed after the extreme weather
165 events. Fish population size at Site-1 changed dramatically after the disturbances (n = 30, $X^2_{(3)}$ =
166 22.43, $p < 0.0001$, Fig. 2). It dropped from, mean ± SD; 193.46 ± 28.13 individuals per 100 m^2,
167 in 2011 before disturbances, to only 48.93 ± 22.84 in 2016 and 23.66 ± 15.69 in 2017 after
168 cyclones and bleaching events. That is, fish population, dropped by 75 % in 2016, and kept
169 declining to reach a loss of up to 88 % in 2017. At Site-2, we also found significant differences
170 in fish abundances between the years (n = 30, $X^2_{(3)}$ = 19.04, $p < 0.0001$, Fig. 2). First, we
171 recorded only a marginal decrease of 26% from, mean ± SD; before: 174.33 ± 130.2 fish
172 individuals per 100 m^2 in 2014, to 129 ± 51.61 in 2016 (i.e., four months after the bleaching
173 event). However, in 2017, a year after the bleaching, we recorded a severe decrease in the overall

fish densities at Site-2, where we counted 37.8 ± 9.26 individuals per 100 m^2, the equivalent of 78 % loss.

By sorting fish communities to the level of a common functional trait value affiliation (i.e., diet composition), we found that the changes in fish densities after the perturbations were not symmetric between Site-1 and Site-2 (Fig. 3). All the statistical outcomes for this section are reported in Table 1. Nine out of 11 functional groups at Site-1 recorded a significant decrease in fish abundance in 2016 and 2017 compared to 2011 fish census data. The nine functional groups that declined in densities at Site-1 were: the browsers, corallivores, detritivores, excavators/scrapers, grazers, macro-invertivores, micro-invertivores, planktivores and spongivores. At Site-2, we only found significant declines in two functional groups: excavators/scrapers already with a decreased abundance in 2016, while spongivores' decrease appeared only in 2017. Planktivores also showed a major decline (Table 1), but the results were statistically not significant. On the other hand, micro-invertivores showed a transient increase in 2016 but declined significantly in 2017, below the densities recorded in 2014. The only functional group that increased in counts at Site-2 in both 2016 and 2017 was the piscivores. Piscivores counts also increased at Site-1 in 2016 and 2017 compared to 2011, but these increases were not statistically significant (see Fig. 3 and Table 1).

Discussion:

We had asked in how far recent extreme weather perturbations such as cyclones 2014/2015 and El Niño 2016, which are predicted to increase as a consequence of climate change, affect coral

reef fish communities at two sites at Lizard Island. Altogether, our findings showed that environmental disturbances correlated with a massive overall loss in fish densities at these sites. Furthermore, the time lag between the perturbation and data collection appeared to be of major importance. In the discussion, we first address the consequences on fish densities at each study site separately, then we discuss general aspects of the present study.

Site-1

Following the 2014 and 2015 cyclones, corals at Site-1 were heavily damaged (Pizzaro et al. 2017). Cyclones usually destroy the reef structure, which would refrain corals from possible rapid recovery (Cheal et al., 2002). The slow recovery of corals might explain fish census in 2016, where we documented substantial declines in fish abundance. Also, almost all fish functional groups at the site suffered severe losses in numbers. The recorded losses support previous findings concluding that habitat structure is essential for fish assemblage (Pratchett et al., 2011). Furthermore, the fish census was even lower in 2017. Currently, we cannot confirm whether habitat degradation due to cyclones kept influencing negatively fish assemblage or whether the 2016 coral bleaching was the driving cause for the declines from 2016 to 2017.

Site-2

Site-2 had been protected from the cyclones by the island. However, it was exposed to coral bleaching in 2016. During the El Niño cycle, in February-April 2016, the Australian Institute of Marine Science reported an increase in seawater temperature by ~2°C above the normal around

220 Lizard Island, reaching thus a maximum of ~31 °C. The same difference was recorded later in
221 June 2016, when the water temperature had dropped down to ~26°C during our fish surveys. The
222 resulting coral bleaching in 2016 was more intense than in previous years, touching up to 60 %
223 of the coral cover at the GBR (Hughes et al., 2017). Such prolonged and strong bleaching would
224 result inevitably in a massive corals death (DiazPulido & McCook, 2002). Losses larger than 10
225 % of the coral cover could be already detrimental to fish populations (Wilson et al., 2006). While
226 our analyses regarding overall fish densities and functional groups yielded statistically non-
227 significant differences between 2014 and 2016 fish censuses, the overall tendency was a decline
228 in fish numbers. Indeed, in a parallel study that explored the effects of the perturbations on the
229 marine cleaning mutualism involving the cleaner wrasse *Labroides dimidiatus* and its 'client'
230 reef fishes, Triki et *al.* (2017) found that the populations of cleaners and client species larger than
231 10 cm were significantly reduced (by 80% and 40%, respectively).
232
233 Importantly, fish census in 2017 showed a severe decrease in fish densities. This delayed
234 decrease in fish densities might be due to lack of food, habitat loss, or both. A potential
235 explanation is that fish surveys in 2016, conducted four months after the onset of El Niño,
236 probably fell in the range of the dying process of the corals due to the bleaching (DiazPulido &
237 McCook, 2002), with their surface not yet over colonised by algae. In contrast to Site-1, major
238 number losses were only recorded in three functional groups: spongivores, excavators and
239 planktivores. Also, two functional groups – microinvertivores and piscivores – showed an initial
240 increase in densities that persisted only in the piscivores group. Probably the bleached corals
241 offered neither suitable shelter for coral-dwelling species (Coker, Pratchett & Munday, 2009;
242 Pratchett et al., 2011) nor a camouflage background, facilitating thus visual recognition of prey

(Phillips et al., 2017). Further monitoring will be needed to test whether two to three years after the perturbation a similar picture will emerge as at Site-1, or whether severe coral bleaching has more variable effects on fish functional groups.

General aspects

The losses in the abundance of some functional groups are detrimental to the reef because most of these functional groups are known for their beneficial role in promoting healthy corals (Green & Bellwood, 2009; Rasher, Hoey & Hay, 2013). In particular, browsers, detritivores, and excavators/scrapers have a diet that is beneficial for coral resilience, coral settlement, and growth, as these groups prevent microalgae from taking over on corals (Green & Bellwood, 2009; Cheal et al., 2010; Rasher, Hoey & Hay, 2013). Equally important are the planktivores and spongivores groups. They reconstitute a large proportion of the overall fish counts on the two studied reef sites (Table 1). The planktivores are an essential functional group for maintaining the ecosystem well equilibrated. In food chains, the planktivores belong to the lower trophic levels in the web. Their role consists of capturing rich nutrients, and transfer them to the bottom-up food chain (Pace et al., 1999; Fisher et al., 2015). Spongivores, on the other hand, have a significant role in protecting corals by feeding on overgrowing sponges, thereby reducing corals vs sponge competition (Hill, 1998). Thus, the recorded loss in spongivores may slow down the speed of coral cover recovery (Hill, 1998).

Overall, most fish functional groups came out as losing because of the environmental perturbations in the long term. Only the piscivores group kept relatively benefiting at both study

sites. Nevertheless, the reduction in prey densities would eventually lead to fewer predators due to the trophic cascade in the food chain.

Conclusion

Our study highlighted the importance of continuous monitoring to assess immediate as well as intermediate-term consequences of extreme weather events. Our findings fit the previously documented negative impact of extreme weather events such as cyclones and El Niño on coral reef ecosystems. It hence appears that such events might have long-lasting adverse effects on fish communities, likely causally linked to the state of the corals that provide shelter to the many small fish species that are at the bottom of the trophic cascade.

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Table 1(on next page)

Summary of the findings in changes of coral reef fish abundance classified into functional groups according to their common shared functional trait.

Bold values indicate statistically significant differences. All statistical analyses are conducted with the non-parametrical Kruskal-Wallis test with *post hoc* analyses ran for significant results only. A sequential Holm-Bonferroni correction was conducted to correct for multiple tests. Based on this correction, the significant threshold α was $\alpha \leq 0.025$ for tests on Site-1 data, and $\alpha \leq 0.007$ for tests on Site-2. Different letter codes indicate significant differences between years within each site in the post hoc analyses.

1

Functional group	Site 1 (cyclones 2014/2015 and coral bleaching 2016 damages)					Site 2 (coral bleaching 2016 damages)				
	χ^2	*P*-Value	Year 2011 (Mean ± SD; rank)	Year 2016 (Mean ± SD; rank)	Year 2017 (Mean ± SD; rank)	χ^2	*P*-Value	Year 2014 (Mean ± SD; rank)	Year 2016 (Mean ± SD; rank)	Year 2017 (Mean ± SD; rank)
Browser	**17.4**	**<0.001**	**1.93 ± 1.67; a**	**0.13 ± 0.42; b**	**0 ± 0; b**	1.0	0.60	0.06 ± 0.21	0 ± 0	0.06 ± 0.21
Corallivore	**11.9**	**0.002**	**2.6 ± 1.67; a**	**1.06 ± 0.78; ab**	**0.33 ± 0.72; b**	2.3	0.31	2.13 ± 1.68	1.26 ± 0.73	1.33 ± 1.75
Detritivore	**19.7**	**<0.0001**	**23.06 ± 10.02; a**	**3.33 ± 2.95; b**	**2.4 ± 2.98; b**	0.8	0.66	4.6 ± 4.8	4.06; 4.34	5 ± 3.62
Excavator/scraper	**19.6**	**<0.0001**	**8.33 ± 4.10; a**	**0.53 ± 0.52; b**	**0.8 ± 1.16; b**	**14.0**	**<0.001**	**6 ± 4.53; a**	**1 ± 1; b**	**1.46 ± 1.62; b**
Grazer	**18.6**	**<0.0001**	**22.6 ± 18.97; a**	**9.33 ± 4.81; b**	**3.13 ± 1.83; c**	6.4	0.040	7.53 ± 5.08	11.26 ± 6.50	5.06 ± 3.16
Macro-invertivore	**11.1**	**0.003**	**6.93 ± 4.04; a**	**4.2 ± 5.99; b**	**1.66 ± 1.67; b**	2.4	0.297	2.86 ± 2.33	2.6 ± 1.67	1.53 ± 1.33
Micro-invertivore	**9.8**	**0.007**	**18.06 ± 6.56; a**	**10.8 ± 6.60; b**	**7.86 ± 6.68; b**	**16.6**	**<0.001**	**21.46 ± 27.53; b**	**34.86 ± 18.23; a**	**7.06 ± 2.16; c**
Pisci-invertivore	1.2	0.540	1 ± 1.51	0.33 ± 0.65	0.53 ± 0.75	4.2	0.118	1.33 ± 1.44	0.4 ± 0.56	0.4 ± 0.64
Piscivore	0.4	0.828	0.6 ± 0.85	0.93 ± 1.26	0.73 ± 0.73	**10.1**	**0.006**	**0 ± 0; b**	**0.86 ± 0.89; a**	**0.46 ± 0.55; a**

Planktivore	**23.7**	**<0.0001**	**64.2 ± 26.11; a**	**2.46 ± 3.22; b**	**0.06 ± 0.21; c**	7.9	0.019	86.99 ± 107.20	27.6 ± 26.32	7.46 ± 5.76
Spongivores	**19.5**	**<0.0001**	**39.73 ± 18.58; a**	**11.06 ± 10.49; b**	**2.73 ± 2.92; c**	**14.1**	**<0.001**	**37.66 ± 22.07; a**	**28.73 ± 16.91; a**	**7.8 ± 3.08; b**

2

Figure 1(on next page)

Study sites with images of their coral cover in May 2015.

(a) Section image from the study of Pizzaro et al. (2017) of Site-1, which is an exposed reef that lost a significant proportion of its coral cover due to the destructive Cyclones Ita in April 2014 and Nathan in March 2015. **(b)** An orthographic image of (a). **(c)** Section image from the study of Pizzaro et al. (2017) of “Horseshoe reef” nearby Site-2 (35 m apart). The reef is inside a protected area from the cyclones. **(d)** An orthographic image (c). **(e)** Lizard Island map is indicating the precise localities of the study sites.

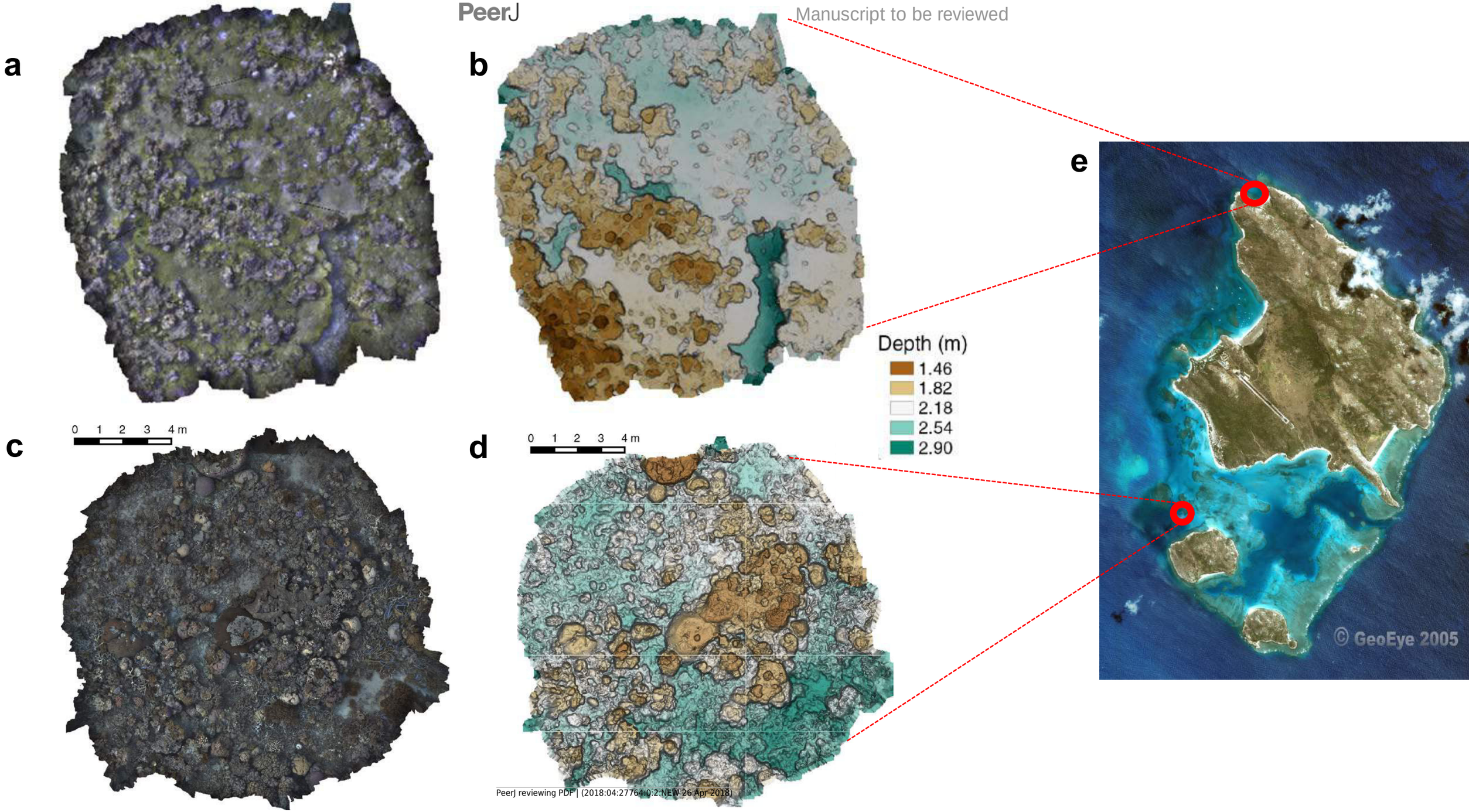
a
b
c
d
e
0 1 2 3 4 m
0 1 2 3 4 m
Depth (m)
1.46
1.82
2.18
2.54
2.90
© GeoEye 2005

Figure 2(on next page)

Fish abundance.

Boxplots are displaying median and interquartile of fish abundance. Different letter codes indicate significant differences between years within each site.

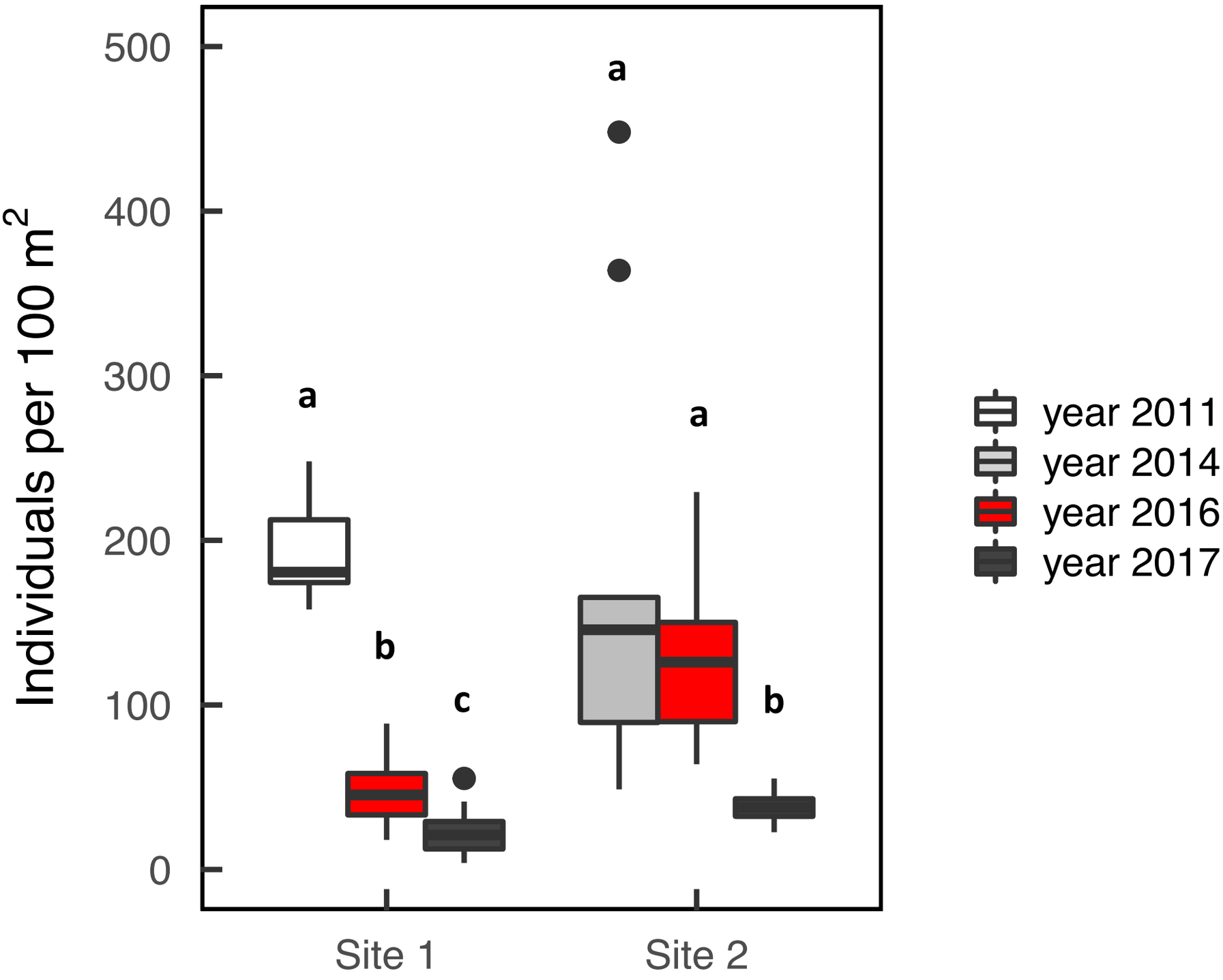
Individuals per 100 m2
500
400
300
200
100
0
a
b
c
a
b
Site 1
Site 2
year 2011
year 2014
year 2016
year 2017

Figure 3(on next page)

Fish abundance in functional groups.

Boxplots are displaying median and interquartile of fish abundance. Note that due to the high variation in fish abundance per functional groups, the y-axes are not similar. See Table 1 for the statistically significant differences in functional groups within reef sites and between years.

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