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# Rapid tourism growth and declining coral reefs in Akumal, Mexico

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**Abstract** Ecosystem-based management efforts in coral reefs typically focus on reducing fishing pressure. However, independent of overfishing, tourism can degrade coral reefs through coastal development, as well as the physical presence of tourists within the ecosystem, the effects of which remain poorly understood. We combined a 3-year dataset on coral and algal cover with a more extensive survey of the benthic community to examine the effect of intensive tourism on a coral reef in Akumal Bay, Mexico. Results from our 3-year dataset indicated that near the peak snorkeling area in the bay, coral cover decreased by 79 % from summer 2011 through summer 2014, a period in which the number of monthly snorkelers increased by more than 400 %. Our summer 2013 survey of the benthic community between sites within a zone of dense snorkeler traffic versus site at a nearby control location revealed negative effects of intensive tourism on particular coral morphologies and on the abundance of herbivorous reef fishes. Our results suggest that uncontrolled tourism, including accelerating growth in the number of snorkelers, is likely contributing to the decline of the coral reef in Akumal Bay, where further expansions in tourism are planned. Indeed, the ecosystems threatened by overexploitation via tourism in the Mayan Riviera also form the basis for the regional tourism industry. Thus, long-term ecological monitoring coupled with the establishment and enforcement of regulations on tourism may be essential for the sustainability of coral reefs, as well as the socioeconomic benefits they provide in Mexico.

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## Introduction

The structure and function of coral reefs center on the balance between corals, upon which the greater community is based, and algae, which compete with corals for benthic substrate (McCook et al. 2001; Smith et al. 2006; Rasher and Hay 2010). Herbivores consume benthic algae and can thus play an integral role in structuring the benthic community (Bellwood et al. 2004; Hughes et al. 2010). Consequently, over-exploitation of herbivorous reef fishes and the resultant release of benthic algae from consumer pressure is believed to be a ubiquitous mechanism through which coral reefs are declining globally (Burkepile and Hay 2006). As a result, ecosystem-based management approaches in coral reefs have focused overwhelmingly on reducing fishing pressure, with little attention being paid to other ecologically threatening human activities (Pandolfi et al. 2005). However, the rising industry of tourism in and around coral reefs can drive coral reef degradation through coastal development and through the presence of tourists within the reef (Hawkins and Roberts 1992; Leujak and Ormond 2008; Uyarra et al. 2009).

It is understood that coastal development can increase the frequency and magnitude of terrestrial runoff events

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that drive nutrient enrichment and sedimentation, both of which can tip the scales in favor of benthic algal domination over corals (Birrell et al. 2005; Fabricius 2005; Muthukrishnan and Fong 2014). However, the ecological role of tourists as visitors within coral reefs is far less understood. While within-reef tourism has the potential to provide an environmentally and economically sustainable alternative to the exploitation of fisheries (Honey 2008; Claudet et al. 2010), snorkel or SCUBA-based reef tourism could also drive ecological degradation through direct, physical contact with reef corals (e.g., touching, trampling, boat anchoring; Harriot et al. 1997; Saphier and Hoffmann 2005; Uyarra et al. 2009; Krieger and Chadwick 2013), re-suspension of sediment that can harm reef corals (Rogers 1990; Birrell et al. 2005), inputs of nutrient-based human waste that promotes algal growth (Fabricius 2005), or modification of the behavior of reef fishes that perceive tourists or boats as threats (Graham and Cooke 2008). Through diverse mechanisms, excessive snorkel or SCUBA-based tourism may undermine the sustainability of both coral reefs and, consequently, tourism industries, upon which many developing nations depend.

Tourism in the Mayan Riviera, Mexico, contributes significantly to the Mexican economy and is growing rapidly. The number of hotel rooms in the state of Quintana Roo (home to the Mayan Riviera) increased by nearly 80,000 between 1975 and 2010 (Baker et al. 2013; INEGI 2014). An increasingly popular tourist destination in this area, particularly for snorkeling, is the town of Akumal (Mayan for "place of the turtle"), which hosts an abundant sea turtle population just offshore in Akumal Bay. Year round, Akumal Bay draws buses and boats full of snorkelers, which grow in number annually. While sea turtles attract snorkelers to Akumal Bay, snorkel tours and the growing tourism infrastructure on land may degrade coral reefs that support these turtle populations, as well as a vast diversity of other marine life. Despite potentially broad socioeconomic consequences, the ecological effects of tourism in Akumal Bay remain largely unknown (but see Roy 2004; Mutchler et al. 2007; Baker et al. 2013).

Snorkel tours concentrate almost exclusively on the north side of Akumal Bay, because this side (1) experiences high densities of green sea turtles, grazing on seagrass habitat; (2) has permanent moorings to guide tour boats into the lagoon; (3) has direct road access for tour buses; and (4) is closer to public restaurants and shops. The coral reef habitat that is just south of this 'snorkeler hotspot' is seldom visited by snorkelers. This explicit spatial pattern in snorkeler use combined with the temporal pattern of increasing tourism in Akumal Bay provides a unique opportunity to address questions regarding the effects of tourism on coral reefs. Here, we report on our in situ investigation of the benthic community in response to differences in tourism over time and space in the coral reef of Akumal Bay.

## Materials and Methods

### Monitoring benthic cover (2011–2014)

Beginning in the summer of 2011, the conservation organization Centro Ecológico Akumal (CEA) extended their long-term monitoring program (began in 2006) to include the backreef of Akumal Bay, where snorkeling has become increasingly popular. Benthic survey data were collected from patch reefs in the backreef using the point-contact method of either 100 points from each of six 10-m-long transects per site (for years 2011–2012, following the AGRRA protocol; Lang et al. 2010) or 120 points from each of five 30-m-long transects per site (for years 2013–2014, following the MBRS protocol; Almada-Villela et al. 2003). These sampling methods yielded 600 substrate contact points, each of which was identified to functional group, providing a measure of relative cover of the benthos for each site for each sampling phase. For each of the 1–3 annual sampling phases, the 5–6 transects were haphazardly deployed within each site, which covered an approximate area of 100 $m^2$. We compared two backreef sites of similar depth (1.4–3.8 and 1–2.2 m, respectively) and environmental conditions within Akumal Bay: one near and one far from the major snorkel tourism traffic zone. The long-term site nearer to the high-tourism area in the bay (LT-near) was monitored from summer 2011 through summer 2014, and the long-term site further from the high-tourism area in the bay (LT-far) was monitored from fall 2012 through summer 2014. In addition, CEA conducted daily monitoring of Akumal Bay snorkeler use from 0800 to 1700 h from January 2011 through the summer of 2014. Time and date of water entry of snorkelers, and the total number of tourists and accompanying guides snorkeling in the bay were recorded by a research assistant aided (via radio communication) by lifeguards located at 2–3 observation points along the shoreline of Akumal Bay.

### Benthic community survey (2013)

In May of 2013, we conducted surveys of benthic taxa, including reef fishes, within patch reef sites subjected to high and low levels of snorkel-based tourism in Akumal Bay, Mexico (denoted as 'Lo' and 'Hi', for low and high tourism sites, respectively, in Fig. 1). We chose our two site categories [i.e., central location: few tourists (control) vs. northern location: many tourists (impact)] based on daily tourist counts (over 7 days) made at 1500 h, a peak hour for tourist activity identified in previous surveys by CEA. Each

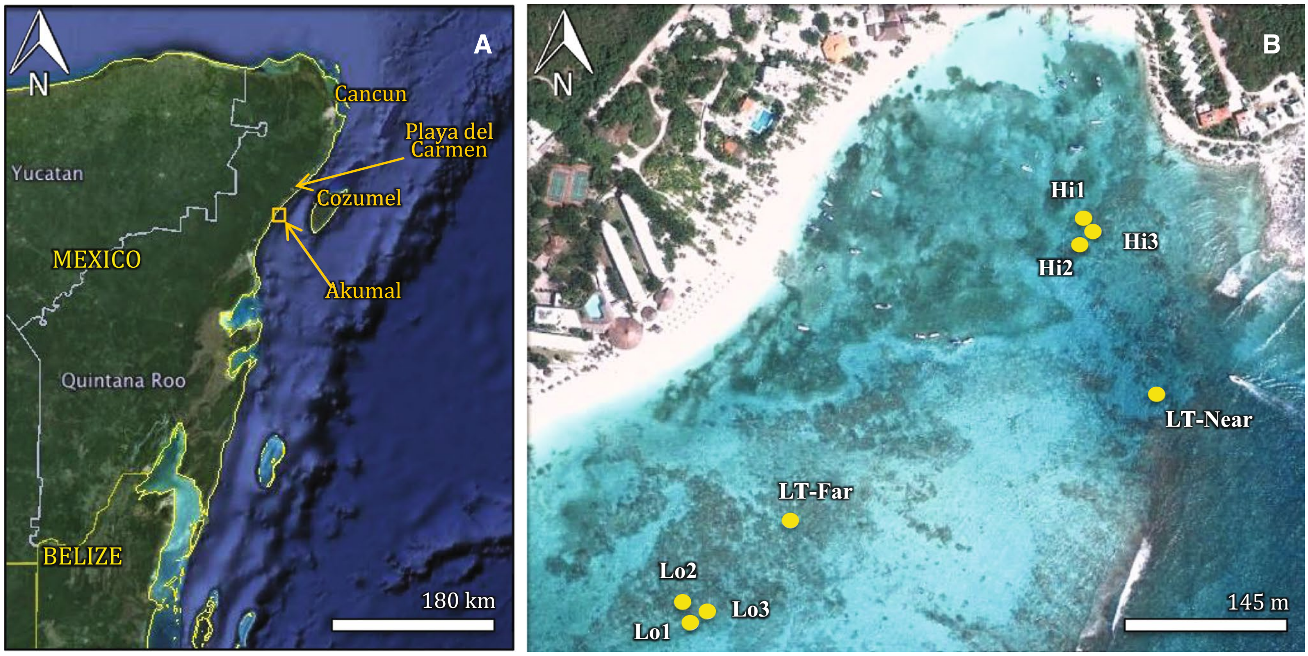


**Fig. 1** Map of our study region (**a**) and study location in Akumal Bay, Mexico (**b**), 20.3945º N × 87.3137º W, including the sites from our multi-year monitoring (LT-far & LT-near) and our 2013 benthic community survey (Lo1, Lo2, Lo3, & Hi1, Hi2, Hi3), where "Lo" designates low-tourism sites and "Hi" designates high tourism sites. The map was created using Google Earth

day, we counted tourists in the water (including snorkelers and swimmers, though the former was the overwhelmingly dominant group) and on the beach by walking (three times) the length of beach parallel to the shoreline from the central to the northern part of Akumal Bay. Within our control and impact site categories, we selected three replicate sites composed of a minimum of 140 $m^2$ of patch reef (hard substrate), over which we conducted transects. We also chose impact and control sites with the least possible differences in depth (1–1.7 and 1–1.2 m) and distance from shoreline (171–188 and 170–189 m, to the west shore), to minimize factors that could confound tourism effects.

At each site, we deployed three 14 m transects parallel to the shoreline. At six randomized points along each transect, we placed a 0.5 $m^2$ quadrat, within which we visually quantified relative benthic cover, including benthic algae, live and dead hard coral, soft coral, and other sessile invertebrates. We also measured the relative abundance of reef fishes using 3 × 14 m belt transects conducted in two replicate trials atop benthic survey transects at each site. We allowed 5 min for fishes to acclimate to the presence of the transect tape before conducting fish counts. Trials were run on different days between the hours of 800–1000 at each site, to control for any differences in fish activity throughout the day. We swam each transect at a fixed pace (0.5 m $min^{-1}$) and counted and identified (at least to family) all fishes present within the 42 $m^2$ area. To minimize the likelihood of double counting fishes between transects, fish counts were conducted simultaneously at all three transects at each site, and thus, three researchers were required. To control for observer bias, the same three researchers conducted simultaneous fish counts across all sites.

## Data analyses

All statistical modeling was conducted in R (R Core Team 2013). For the long-term monitoring data on benthic cover, we analyzed the response of coral cover and algal cover to the main effects of site and time, as well as the interaction between these two factors, using linear models. We log-transformed the coral cover data [log(x + 1)] to meet model assumptions of normality and homoscedasticity. For long-term snorkel-based tourist monitoring, we analyzed the effect of time on monthly tourist numbers using a linear model with a quadratic term. For data from the 2013 benthic community survey, we used linear models to compare daily tourist counts between our two site types, and we used linear mixed effects models within the R package nlme (Pinheiro et al. 2013), to measure the response of corals, algae, and fishes to snorkel-based tourism (impact vs. control; fixed effect), treating study site and transect as nested random effects. Data were transformed and/or weighted based on variance, as needed, to meet model assumptions of normality and homoscedasticity (see Table 1 for data transformations and variance weighting functions used).

**Table 1** Summary of data analyses performed in R (R Core Team 2013) on data from our multi-year monitoring and benthic community survey in Akumal Bay

| Approach | Response | Factor | Coefficient | $p$ | Significance |
|---|---|---|---|---|---|
| Multi-year monitoring | Hard coral cover* | Time (mo.) | −0.366 | <0.001 | *** |
| | ('near' site only; all dates) | Snorkelers | −0.000068 | 0.013 | * |
| | Algal cover | Time (mo.) | 0.42 | 0.074 | . |
| | ('near' site only; all dates) | Snorkelers | 0.00078 | 0.12 | |
| | Hard coral cover* | Site | −0.53 | 0.0093 | ** |
| | ('near' and 'far' sites; | Time (mo.) | 0.0059 | 0.38 | |
| | fall 2012 to fall 2014) | Site/time | −0.033 | 0.17 | |
| | Algal cover | Site | 1.53 | 0.78 | |
| | ('near' and 'far' sites; | Time (mo.) | 0.76 | 0.023 | * |
| | fall 2012 to fall 2014) | Site/time | 0.2 | 0.77 | |
| Benthic community survey | No. of tourists in water* | Site | 2.46 | <0.0001 | *** |
| | No. of tourists on beach* | Site | 1.00 | 0.020 | * |
| | Branching coral cover†Δ | Site | −0.84 | 0.068 | . |
| | Mounding coral coverΔ | Site | −3 | 0.47 | |
| | Plating coral cover^Δ | Site | −7.6 | 0.014 | * |
| | Dead coral cover‡Δ | Site | 0.19 | 0.039 | * |
| | Algal cover†Δ | Site | 0.21 | 0.64 | |
| | No. of macroalgal spp.*Δ | Site | 0.11 | 0.18 | |
| | No. of fishes*Δ | Site | −0.35 | 0.19 | |
| | No. of fish families | Site | 0.28 | 0.67 | |
| | No. of herbivorous fishes*^ | Site | −0.52 | 0.052 | . |

Monitoring data were analyzed using linear models (fixed effects only), and benthic community survey data were analyzed using linear mixed effects models, treating site and transect as nested random effects (Pinheiro et al. 2013)

Significance codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1

* Data were log-transformed; † Data were square-root transformed; ‡ Data were arcsine square-root transformed; ^ Variance was modeled with an exponential function (Pinheiro et al. 2013); Δ Variance was modeled with a power function (Pinheiro et al. 2013)

## Results

All statistical models met assumptions of normality and homoscedasticity, as verified by model residual plots. The number of snorkelers in Akumal Bay grew at an accelerating rate over a period encompassing our study ($r^2 = 0.71$, $p < 0.0001$): monthly snorkelers increased over fourfold in 3 years, peaking at <5000 in the summer of 2011 to over 20,000 in the summer of 2014 (Fig. 2). The results from our multi-year monitoring data of the site nearer to the high-tourism area of the bay, from summer 2011 through summer 2014, showed that time had a significant negative effect on coral cover ($r^2 = 0.29$, $p < 0.001$), with an average loss of 79 %, from 16.7 to 3.5 % coral cover (Fig. 3). Time had a marginally significant positive effect on algal cover ($r^2 = 0.086$, $p = 0.076$; Fig. 3). Similarly, the number of snorkelers for each benthic survey month had a significant negative effect on coral cover ($r^2 = 0.16$, $p = 0.01$) and a positive, albeit insignificant, effect on algal cover ($r^2 = 0.064$, $p = 0.1$). Regarding our multi-year monitoring

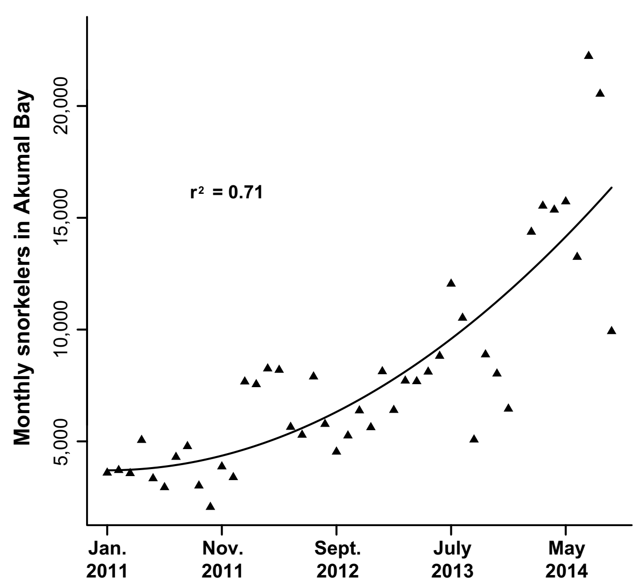


**Fig. 2** The number of monthly snorkelers visiting north Akumal Bay over our study period

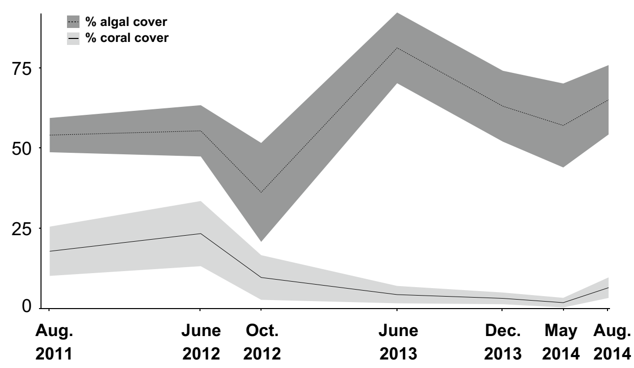


**Fig. 3** The mean (±95 % CI) percentage of substrate covered in coral (*solid line*, *light gray error bars*) and benthic algae (*dotted line*, *dark gray error bars*) from summer 2011 through summer 2014 from random transects taken from a site near the major snorkel tourism traffic zone in north Akumal Bay

data for both high and low (control) tourism sites (fall 2012 through summer 2014), for coral cover, the main effect of site was significant ($p = 0.009$), with the location nearer to the snorkeling hotspot having lower coral cover (Fig. 4). The main effect of time on coral cover was insignificant ($p = 0.4$). Conversely, for algal cover, the main effect of site was insignificant ($p = 0.8$), while the main effect of time was significant ($p = 0.03$), indicating an average annual increase in algal cover of 9.1 % in Akumal Bay (Fig. 4). There was no significant interaction between site and time for coral ($p = 0.2$) or algal cover ($p = 0.8$), indicating that changes in coral or algal cover over time did not differ significantly between the sites for the 2 years that they were both monitored (from fall 2012 through summer 2014).

Results from our summer 2013 benthic community survey complemented our monitoring data by providing more specific insights regarding benthic communities between control and impact sites (summarized in Fig. 5; Table 1). First, our daily tourist counts validated our site category assignments, indicating significantly greater numbers of tourists in the water (13-fold greater; $p < 0.001$) and on the beach (3.5-fold greater; $p = 0.020$) for the impact relative to the control location. We observed a negative effect of high tourism on hard coral cover for all coral morphologies; though this effect was insignificant for mounding corals [37.3 % decrease (from control to tourism sites), from 8.94 to 5.61 % cover; $p = 0.5$], it was marginally significant for branching corals (90.4 % decrease, from 1.26 to 0.12 % cover; $p = 0.07$) and significant for plating corals (93.4 % decrease, from 8.1 to 0.54 %; $p = 0.01$). High-tourism locations also had significantly higher dead coral cover (50.5 % increase, from 38.1 to 57.4 %; $p = 0.04$) and a higher abundance (15.2 % increase, from 17.1 to 19.7 % cover) and number of taxa (77 % increase, from 24.6 to 43.5 taxa) of benthic algae, though these latter effects were

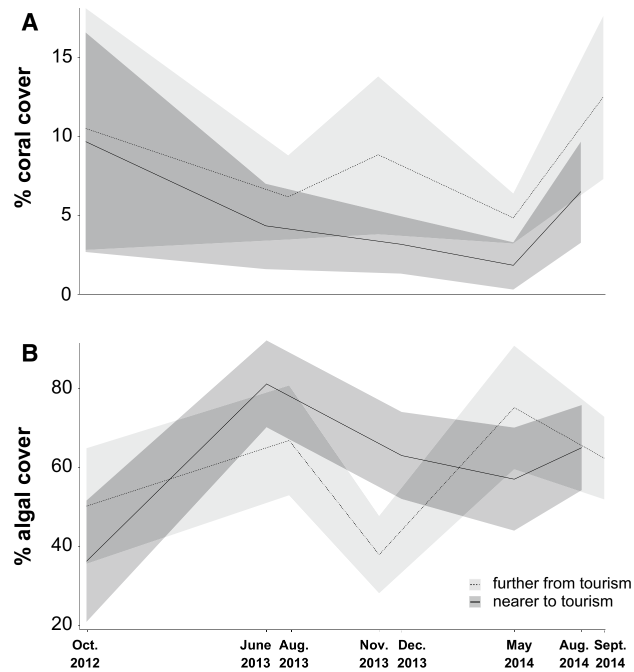


**Fig. 4** The mean (±95 % CI) percentage of substrate covered in coral (**a**) and benthic algae (**b**) from fall 2012 through summer 2014 from random transects taken from a near site (*solid line*, *dark gray error bars*) and a far site (*dotted line*, *light gray error bars*) relative to the major snorkel tourism traffic zone in north Akumal Bay

insignificant ($p = 0.6$ and 0.2, respectively). Finally, high tourism had insignificant effects on the abundance of reef fishes and the number of reef fish families but a marginally significant negative effect on the abundance of herbivorous reef fishes (73.9 % decrease, from 41.0 to 23.6 fishes; $p = 0.05$).

## Discussion

Our data establish a link between tourism and declines in coral cover over time (Figs. 2, 3; Table 1) and space (Fig. 5) in Akumal Bay, Mexico. In particular, our data suggest that snorkel-based tourism in the bay may reduce coral cover, though the magnitude of this effect was trait dependent, with more structurally reinforced mounding morphologies of coral showing greater resilience than more fragile branching or plating morphologies (Fig. 5). Thus, tourism within coral reefs may affect the species composition of corals, which serve different functional roles in the system and could, consequently, have different feedbacks on other members of the community (e.g., fishes and invertebrates that reside in/under branching/plating corals). Moreover, our data indicate that fish abundance (particularly that of herbivores; Fig. 5) is also negatively affected by snorkel-based tourism, suggesting that the accelerating growth

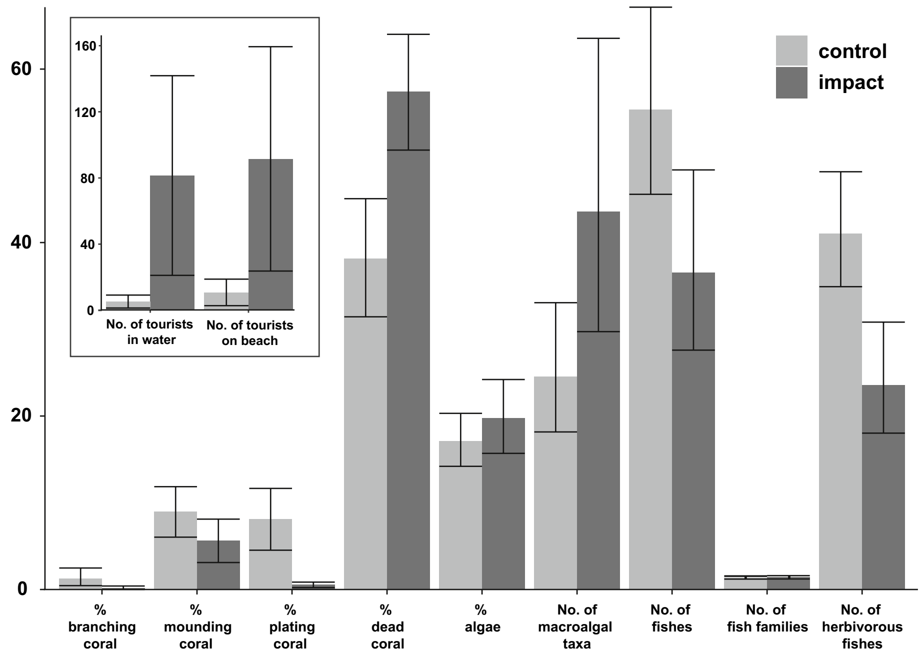


**Fig. 5** Characteristics of corals, algae, and fishes from the low-tourism "control" sites (*light bars*) and the high-tourism "impact" sites (*dark bars* located inside the major snorkel tourism traffic area in north Akumal Bay) in May 2013. Reported mean (±95 % CI) percentages and algal taxa counts come from back-transformed data from relative cover estimates from quadrats replicated within transects replicated at each of three control and impact sites. Reported mean (±95 % CI) fish counts come from back-transformed data (see Table 1) from replicated visual belts over these same transects

rate of snorkelers in Akumal Bay (Fig. 2) could elicit top-down (consumer-mediated) controls on the composition of the benthic community indirectly, through reef fish habitat loss (i.e., coral declines), or directly, through trait-mediated effects on herbivorous fishes (Preisser et al. 2005; Madin et al. 2010a, b). These effects could occur in combination with bottom-up effects (e.g., nutrient pollution, sedimentation) that favor the growth of benthic algae that increased in Akumal Bay over time (Fig. 4) and that threaten corals and the stability of the greater ecosystem (Bellwood et al. 2004).

Several mechanisms could have driven the patterns we observed in our data, and though our site categories of low and high tourism were inherently clustered in space (Fig. 1), we expect confounding factors to have played a minor role in our observed effects of tourism. First, while coastal development can enhance nutrient runoff, this is unlikely to have driven differences between our high tourism and control sites, because Akumal Bay is situated atop a karstic groundwater system, which diffuses wastewater widely (Mutchler et al. 2007; Nicholls 2008; Baker et al. 2013). For example, Baker et al. (2013) showed a correlation between levels of regional tourism and excess nitrogen accumulated in the tissues of sea fans >1 km offshore, on the forereef outside of Akumal Bay. Thus, if nutrients of terrestrial origin affected our study sites, they likely had similar effects across sites, which were proximate in space (Fig. 1). Moreover, large-scale nutrient runoff effects could explain the general increase over time of algal cover in the bay (Figs. 2, 3), as well as the lack of significant site differences (multi-year monitoring sites) regarding changes in coral or algal cover over time (Fig. 4; Table 1). Conversely, snorkel tourists directly deposit nutrient waste into the reef through urination, which could potentially lead to localized spikes in nutrients that could facilitate algal production (though insignificant increases in algal abundance and richness provided weak evidence for this effect; Fig. 5) or inhibit coral growth directly (Fabricius 2005; Gil 2013).

Differences in water flow and storm surge between our sites could have contributed to the patterns in our responses, though we expect that these contributions did not bias our results. First, all of our study sites were generally close in space and at similar depths within a tidally protected backreef location with relatively calm wave action (Fig. 1). Nonetheless, water flow from wave action in calm backreef habitats can dissipate significantly with distance from the reef crest (Hench et al. 2008), and while our 2013 community survey sites were similarly distant from the reef crest, our multi-year survey site near the high-tourism zone of the bay was positioned closer to the reef crest than our other long-term site (Fig. 1). Because increased water flow can enhance coral growth and survival and reduce harmful effects of sediment, nutrients or algae on corals (Nakamura and van Woesik 2001; Brown and Carpenter 2013; Comeau et al. 2014; Gowan et al. 2014), we would expect flow to increase coral cover at our multi-year survey site near the high-tourism zone relative to the control site (further from the reef crest), a pattern that opposes what we observed (Fig. 4). Thus, it is unlikely that flow confounded the effect of tourism on our multi-year monitoring data. In addition, while storm surge can drive pronounced increases in flow that could affect parts of Akumal Bay differently (e.g., the south vs. east-facing shore, Fig. 1), over our study period only one major storm system (tropical storm Rina) passed over Akumal (on October 28, 2011), but did not elicit observable changes in either coral or algal cover (Fig. 3). While differences in flow due to oceanographic and meteorological processes were unlikely to have qualitatively affected our results, snorkelers can have localized effects on water flow that cause re-suspension of sediments into the water column. Resultant increases in sedimentation can reduce coral survival and recruitment and also facilitate algal overgrowth (Rogers 1990; Birrell et al. 2005). Moreover, physical disturbances from snorkelers incidentally coming into contact with corals can damage coral tissue and skeletons and thus could have caused observed increases in dead coral cover (Fig. 5; Hawkins and Roberts 1992; Krieger and Chadwick 2013).

While fishing reduces fish abundance and species richness in many coral reef systems (Bellwood et al. 2004; Hughes et al. 2010), the fishing industry in Akumal Bay is almost exclusively an offshore enterprise, due, at least in part, to competition with lucrative and dominant snorkel-based tour operations. However, non-fishing snorkelers may nonetheless affect the behavior and thus the local abundances of fishes (Di Franco et al. 2013), which may perceive Akumal's snorkel tourists (which generally do not feed fish) as predators. Indeed, we observed fewer reef fishes (particularly herbivores) at high snorkel-based tourism sites, an effect that could be driven by trait-mediated effects of snorkel tourists (Preisser et al. 2005). However, reef fish declines in sites with high snorkel tourism could also be driven by associated declines in corals (Fig. 5), which provide both refuge and settlement cues to reef fishes (Paddack et al. 2009; Dixson et al. 2014). No matter the mechanism, reductions in the abundance or foraging behavior of herbivorous reef fishes could release algae from grazing pressure (Preisser et al. 2005), the effects of which we may have only begun to observe in our data, showing positive, though insignificant, effects of intensive tourism on the abundance and richness of algae (Fig. 5; Table 1). Furthermore, recent work has shown that the combination of reduced herbivore densities and high sedimentation (which can be driven by snorkelers) can have synergistic negative effects on coral cover (Muthukrishnan and Fong 2014).

Tourism in Akumal remains entirely unregulated, and there are imminent plans to expand the tourism enterprise, including hotels and snorkel tours, to the south side of Akumal Bay. This pattern of development is characteristic of the greater region of the Mayan Riviera, where tourism continues to grow and expand rapidly, now reaching far south of Cancun toward the Belize border (Rioja-Nieto and Sheppard 2008; Figueroa-Zavala et al. 2015). Tourism based on ecological attractions is an essential component of the economy in many developing nations (Honey 2008) such as Mexico, where tourism in Quintana Roo forms the backbone of the economy. However, our data suggest that rapidly growing tourism in this region may be unsustainable, as tourism in Akumal Bay is linked to ecological degradation of the valuable coral reef ecosystem. This ecological degradation can not only have adverse effects on various resident organisms in the reef system (including charismatic sea turtles) but can also have important feedbacks on tourism itself. For example, studies show that reef tourists place the greatest value on coral cover, complexity and fish abundance (Uyarra et al. 2009), which show negative responses to tourism in Akumal Bay (Figs. 3, 5). Additionally, proximate environmental effects of tourism could interact with ultimate stressors brought on by global climate change, potentially further increasing the susceptibility of coastal tourism hotspots to costly ecological degradation (Bridge et al. 2014; Foster et al. 2014). For these reasons, it is imperative that long-term monitoring efforts in Akumal Bay be continued and expanded to include additional locations throughout the bay and in nearby sister bays. The insights gained from monitoring and other scientific efforts can be used to inform tourism regulations, with the goal of ecological preservation for long-term socio-economic sustainability.

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