Diving tourism on coral reefs in Florida:
Variation in recreational diver behavior and impacts on reef corals

by

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Abstract

Ecotourism often is promoted as an ecologically-friendly use of natural resources, however if not managed carefully, can cause negative impacts to ecosystems. In the Florida Keys, minimal government involvement in regulating recreational diver behavior has led to local dive shops being mainly responsible for promoting diver behavior that reduces damage to the reef ecosystem. I determined patterns of recreational dive frequency and damage to reef corals at selected sites near Key Largo, and analyzed the effects of pre-dive briefings and other factors on diver behavior. Field observations revealed about 18 diver-coral contacts per each 60-min scuba dive. The majority of coral colonies were damaged, and live coral cover was as low as 11% at frequently-visited reef sites, indicating that current rates of recreational diving on some reefs in Key Largo appear to be ecologically unsustainable. This study reveals severe ecological consequences of current management practices of diving tourism in Florida.

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CHAPTER 1

Recreational diving tourism: impacts and implications for coral reefs

1. Types of ecotourism

Ecotourism has been defined as “responsible travel to fragile, pristine, and usually protected areas that strive to be low impact and small scale” (Honey 1999). Its purpose is to “educate the traveler, provide funds for ecological conservation, directly benefit the economic development and political empowerment of local communities, and foster respect for different cultures and for human rights” (Honey 1999). Ecotourism typically focuses on volunteering and environmental responsibility. It generally centers on travel to destinations where flora and fauna are the primary attractions. A main goal of ecotourism is to offer tourists a look into the impact of human beings on the environment, and to inspire a greater understanding and respect for natural habitats. Therefore, in addition to evaluation of natural environments, an integral part of ecotourism is the promotion of recycling, energy efficiency, water conservation, and creation of economic opportunities for local communities (Randall 1987). For these reasons, ecotourism often appeals to environmental and social responsibility organizations.

However, within the diversity of activities that range from conventional tourism to ecotourism, there has been much disagreement as to the levels of biodiversity preservation and environmental impact that can be included in ecotourism (Buckley 1994). For this reason, environmentalists, special interest groups, and governments define ecotourism differently. Nature tourism, low impact tourism, green tourism, bio-tourism, ecologically responsible tourism, and others terms have been used in the literature and in marketing, although they are not necessary synonymous with ecotourism (Tuohino and Hynonen 2001). Environmental organizations have generally insisted that ecotourism is conservation supporting, sustainably managed, nature-based, and environmentally educated (Buckley 1994, Tuohino and Hynonen 2001). However, the tourist industry and many governments focus primarily on the product aspect, treating ecotourism as any sort tourism based in nature (Tuohino and Hynonen 2001).

2. Impacts of ecotourism on terrestrial trail systems

Although ecotourism is one of the fastest-growing sectors of the tourism industry, growing annually by 10-15% worldwide (Weaver and Lawton 2007), many ecotourism projects fail to meet self proclaimed standards of integrity. While ecotourists claim to be educationally motivated and environmentally concerned, they rarely understand the ecological consequences of their visits and how their day-to-day activities impact the environment (Wearing and Neil 2000).

Ecotourism activities are categorized as environmental impacts because they disturb flora and fauna. Many ecotourists believe that because they are taking only pictures and leaving only footprints, they keep ecotourism sites pristine, but even seemingly harmless activities such as nature hiking and trail walking can be ecologically destructive. These trails serve as a direct link between tourists and the environment, and ultimately serve to concentrate adverse impacts such as trail trampling (Symmonds et al. 2000). In the Annapurna Circuit in Nepal, ecotourists have worn down the marked trails and expanded to created alternate routes, contributing to soil impaction, erosion,fragmentation and plant damage (Tuohino and Hynonen 2001). In the Jiuzhaigou Biosphere Reserve in China, trail widening by tourists has also become a cause for concern. As these trails become more worn and degraded, people tend to move closer and closer to the edge of the trail, nearer to undamaged habitat. This results in further trampling and expansion of the trail (Li et al. 2005). Where ecotourism activities involve wildlife viewing, they may cause displacement of animals, disrupt feeding and nesting sites, or possibly acclimate them to the presence of humans. In Kenya, wildlife-observer disruption drives cheetahs off their reserves, increasing the risk of inbreeding, illegal poaching, and further endangering the species (Tuohino and Hynonen 2001).

3. Marine ecotourism and threats to coral reefs

Terrestrial environments are not the only venues for ecotourism-based activities. Marine ecotourism is an equally large, if not more extensive activity, in which governments and other organizations have tried to promote interactions between humans and the environment (Garrod 2004). Ecotourism has been promoted especially on coral reefs which are among the most beautiful and ecologically diverse ecosystems in the world. They occur predominately in tropical and subtropical coastal waters, providing a complex and unique environment for millions of organisms. Coral reefs are also a vital economic and commercial resource that has been over utilized and undermanaged in many countries. Due to their locations at appealing sites with tropical allure, coral reef ecosystems attract millions of visitors annually, contributing to their rapid decline (McClanahan 1999). It has been estimated that as much as 5% of worldwide live coral cover has already been lost (Wilkinson 2004) and that some areas have lost as much a 80% live coral cover as a consequence of both natural and anthropogenic influences (Gardner et al. 2003). If current human impacts are left unchecked, coral reef ecosystems are in danger of global collapse.

Fortunately, coral reefs are adapted for various types of physical disturbances and environmental stressors which not only enhance their resilience to change but also make them an ideal system for documentation of effects of ecotourism. Coral reefs experience periodic stresses from many natural factors including storms that cause coral fragmentation and abrasion. River and land sediment run-off also cause stress to reef ecosystems by reducing water clarity and the amount of light available to the photosynthetic algae that are symbiotic in coral tissues. Increased sedimentation on the coral reef surface can lead to reduced rates of feeding and recruitment by corals (Rogers 1990, Reboton and Calumpong 2000). Naturally occurring coral diseases including black band and white pox have also been responsible for killing corals, especially in the Caribbean Sea (Goreau et al. 1998). Various corallivorous fishes such as parrot fish (*Sparisoma*) bite off chunks of coral and grind them in their mouths. While feeding on corals they cause damage by predation and have been observed to bang into corals, causing them to fracture (Miller and Hay 1998). One of the most widely-publicized examples of biological stress to coral reefs is predation by the crown-of-thorns sea star *Acanthaster planci*. This sea star feeds on the tiny coral polyps which comprise reefs, and has had a tremendous impact on areas of the Great Barrier Reef, reducing percent coral cover from 16-40% on areas with no history of *Acanthaster* to coverage of only 6-9% in affected areas (Lourey et al. 2000). As coral systems are able to withstand such a barrage of natural stressors, it is possible to assume that introducing "eco-friendly" managed businesses and tourist activities would result in relatively inconsequential negative impacts. Unfortunately this is generally not the case.

Coastal development has had severe negative impacts on coral reef ecosystems. Human population increases near coral reefs cause pollutants to enter local water systems. Increased river-runoff that contains chemicals (nitrogen and phosphorous from fertilizer) and sediment (from exposed soils) have resulted in increases in coral bleaching events. These processes have contributed 87% of the corals in inshore areas of the Great Barrier Reef becoming bleached to at least some extent, compared to 28% in offshore areas (Berkelmans and Oliver 1999). Increases in river-runoff pollutants also parallel increases in algal biomass along coastal areas. Some macroalgae compete for the hard substrate that new corals need to colonize (Chadwick and Morrow 2011). These algae out-compete colonizing corals, thus limiting the reef's ability to grow and recover from damage (Birrell et al. 2005).

Areas with established coastal infrastructure have experienced dramatic increases in visitors over the few past decades (Divisekera 2003). This has led to the demand for more beach area and facilities to support already established tourist infrastructure. To make room, additional coastline areas are cleared (including mangrove areas) resulting in an increase in sediment run-off from the mainland (Fabricius 2005). With a loss of coastal vegetation, an increased amount of sediment is washed out to sea during rains, causing stress to reef ecosystems by reducing water clarity.

4. Recreational diving effects on coral reefs

Recreational scuba diving is the most direct way that humans physically interact with coral reefs. Because of this, regulation of diver behavior on reefs is of the utmost importance. Although scuba diving is promoted as a true "ecotourist" activity, recreational divers have already caused extensive negative impacts on coral reef ecosystems. Divers in the Eilat, Israeli Red Sea contact live coral substrate as many as 25 times during a 45 minute dive (Zakai and Chadwick-Furman 2002). While this number may seem small by itself, when this contact rate is multiplied with the 250,000-300,000 divers that visit this reef area in a single year, the resulting contact rate per reef area is staggering (Zakai and Chadwick-Furman 2002). While each contact between diver and coral does not necessarily lead to the death of the coral, it contributes to an overall deterioration in coral health (Hawkins et al. 1999).

The majority (as much as 80%) of diver-coral contacts are unintentional (Barker and Roberts 2004). Such contacts typically arise from the inexperience of the diver and/ or a lack of understanding that their actions can cause harm to corals (Barker and Roberts 2004). Addressing this issue is the responsibility of the local governments and dive agencies or businesses that establish the facilities and programs whereby divers are able to visit these reefs.

Efforts by coral reef managers to reduce diver-coral contact have produced mixed results. The most common form of contact prevention is short briefings that are given by dive shop personnel in an effort to raise reef awareness. These briefings were designed to teach proper reef etiquette to divers and to help reduce unintentional contacts. However, not all management facilities administer these briefings and of those that do, many vary in length and detail and have varying effects on diver behavior (Barker and Roberts 2004). The greatest deterrent to diver damage appears to be direct intervention by underwater dive guides, which reduces diver-coral contact rates from 11.6 to 2.4 contacts per 40-min dive (Barker and Roberts 2004). Studies examining the impacts of divers on corals have been conducted on several coral reef regions including the Great Barrier Reef in Australia (Rouphael and Inglis 1997, Harriott et al. 1997), the northern Red Sea (Hawkins and Roberts 1993, Zakai and Chadwick-Furman 2002), and the Mesoamerican Reef in the Caribbean Sea (Gibson et al. 1998). Many have addressed how proper reef management, through the use of dive briefings and/ or dive guides, can reduce diver-coral interactions and thus improve reef quality.

5. Recreational diving impacts in Florida

Despite widespread interest in coral reef conservation worldwide, few studies have focused on diver impacts to reefs in the Florida Keys. The earliest publication on recreational diver impact in Florida was almost 30 years ago on reefs in Biscayne National Park (Tilmant & Schmahl 1981). The authors concluded that recreational diving was a good source of income for the area, and that parks should encourage more divers because there was no apparent harm to the reefs. This conclusion was made during a period when diving rates were possibly low enough to cause little damage, so it is not surprising that it differs from the conclusions of more recent studies. Little information currently exists on the Florida Keys to challenge these 30-year-old findings. Talge (1992) found that 4-6% of total live coral cover in Florida was touched by divers on a weekly basis, and concluded that this did not cause permanent harm to the corals, in contrast to more recent findings on coral reefs worldwide (Plathong et al. 2000, Zakai and Chadwick-Furman 2002, Barker and Roberts 2004). A recent study by Camp and Fraser (2012) found that divers in the Key Largo area exhibit behaviors that negatively impact coral reefs, and that various environmental education tools and strategies could be effective in mitigating these diver-coral contacts. However, no published studies in nearly two decades have determined variation in amounts of broken and sedimented corals on reefs in Florida, and related them to estimated frequencies of recreational diving on each reef. With recent global increases in the climatic, biological, and human-induced factors affecting coral reefs, an updated analysis of recreational diver impacts on Florida reef corals is needed.

The present study is among first to address connections between dive shop management practices and diver-induced coral damage on a wide range of coral reefs in the Florida Keys. Similar studies conducted around the world have shown that briefings provided by dive shops and tour guides are among the few management practices that may significantly reduce diver-coral contact rates (Medio et al. 1997, Plathong et al. 2000). A recent study at 3 reef sites in the Florida Keys came to a similar conclusions (Camp and Fraser 2012). However, coral communities vary widely in characteristics such as the proportion of branching corals, angle of reef slope, etc., and thus also vary in the types and rates of damage caused by recreational divers (Rouphael and Inglis 1997, Plathong et al. 2000). Therefore, a broader study at reef sites spanning the length of the Key Largo area is needed to provide information on reef conditions and diver management issues throughout the northern Florida Keys.

In an effort to assess current coral reef management practices and their ecological impacts in Key Largo Florida, this project addresses two major types of questions concerning the effects of recreational divers on coral reefs:

1) How do the diver educational practices of Key Largo dive shops affect recreational diver contacts with live hard corals? Which types of current practices are ineffective, and how can dive shop resources be diverted to techniques that are more effective?

2) What types of physical impacts do recreational divers have on hard coral distribution, abundance, and physical condition on Key Largo reefs?

CHAPTER 2

Impacts of recreational diving on coral reef at Key Largo, Florida

1. Introduction

Coral reef ecosystems are among the most beautiful and ecologically diverse habitats in the world. They occur predominately in tropical and subtropical waters, providing a complex and unique environment for millions of organisms. Coral reefs also are a vital economic and commercial resource that has been over utilized and undermanaged in many countries (Arin and Kramer 2002). On average, 5% of worldwide coral cover has already been lost (Wilkinson 2004), with some areas experiencing as much as 80% loss in coral cover (Gardner 2003). Coral reef ecosystems continue to be impacted by anthropogenic and environmental stressors, and are in danger of global collapse.

As coral reefs become more accessible and methods for reaching them improve, the number of people diving on reefs is increasing (Hawkins and Roberts 1993). In 1981, an estimated 10,000 scuba divers visited reefs in Biscayne National Park near Miami, Florida, USA (Tilmant and Schmahl 1981). At that time, this number was deemed an accurate representation of diver frequency on similar reefs throughout the world. Twenty years later, 250,000-300,000 divers per year visited reefs near Eilat, Israel, accounting for an estimated 400,000 instances of coral damage annually by divers (Zakai and Chadwick-Furman 2002). With this large increase in recreational diving has come a parallel increase in physical contact between divers and corals, resulting in reef damage mostly caused by diver inexperience and/or ignorance (Davis and Tisdell 1995, Barker and Roberts 2004). Acts of diver damage include contacting corals with fins or gear, kneeling on or grabbing corals, and kicking sediment onto corals (Rouphael and Inglis 1997).

Recreational diving causes both direct (physical contact) and/or indirect (sediment deposition) damage to stony corals. Physical contact with corals often results in coral tissue abrasion and/or skeletal fragmentation (Hawkins 1999). Corals possess exoskeletons of calcified material that provides structural support underneath a thin layer of living tissue on the outer surface (Barnes 1980, Furla 2000). Physical pressure on this tissue layer results in abrasion (wound formation and crushing of skeletal elements) and tissue removal, and may even fracture large sections of the skeleton. Abraded corals are more susceptible to predation and disease, resulting in death of the coral (Rosenburg et al. 2007, Guzner et al. 2010). Coral fragmentation occurs when a piece of skeleton is broken off the colony and settles to the sea floor, where it may become covered by sediment and starved of light and nutrients (Hawkins and Roberts 1999). Fragmented corals may cease sexual reproduction and divert their energy toward re-growing the lost skeleton and tissue of the colony (Zakai et al. 2000). Sedimentation on the reef also occurs when divers swimming nearby kick up sand and other particles from the surrounding soft substrate, covering the reefs and inhibiting coral photosynthesis, recruitment and feeding (Hasler and Ott 2008). Sedimentation rates onto live corals can be high in the areas immediately surrounding mooring buoys and entrances to underwater trails, which divers frequent (Hawkins and Roberts 1993, Hasler and Ott 2008).

Efforts by coral reef managers to reduce diver-coral contacts have produced mixed results. Prior to divers entering the water, short briefings are sometimes given by dive shop personnel in an effort to raise reef awareness and discourage diver-coral contacts. These briefings have a varying effect on diver behavior, depending on their content and length (Barker and Roberts 2004). The most successful deterrent to diver damage appears to be direct intervention by underwater dive guides, which can reduce diver-coral contact rates by as much as 80% (Barker and Roberts 2004).

Frequencies of recreational diving on coral reefs in Florida appear to be some of the highest in the world. In response to a prohibition of anchoring on Florida reefs in 1998, the Florida Park systems increased the number of mooring buoys located around popular reefs, thus increasing the number of boats that can be moored at each reef at any given time (Causey 2002). While buoy use likely has reduced anchor damage to reefs, it also has led to elevated numbers of divers in the water around popular reefs (pers. comm. John Pennekamp State Park officials), and also likely has resulted increased diver-coral contacts. Due to minimal governmental involvement in regulating recreational diver behavior in the Florida Keys, local dive shops are heavily responsible for promoting diver awareness of reef conservation (pers. comm., Scott Donahue, Science Coordinator, National Oceanic and Atmospheric Association [NOAA]), in an attempt to reduce diver-coral contacts.

The present study examined educational practices of dive shops and their effectiveness in reducing reef coral damage in Key Largo, Florida. I quantified variation in the coral contact behaviors of recreational divers, their effect on stony corals, and coral condition on reef systems exposed to differing rates of recreational diving.

2. Methods

2.1. Selection of study sites and dive shops

This study was conducted during May-August 2011 on coral reefs near Key Largo, in the Florida Keys National Marine Sanctuary (FKMNS, Fig. 1), Florida, USA. Study sites were selected that each consisted of a patch reef with a reef flat at 4-13 m depth, to control for effects of varying topography on diver behavior and reef damage (Hawkins and Roberts 1993, Rouphael and Inglis 1997). Due to the topography of the Florida Keys, all sites were located ~ 5-7 miles (8-11 km) offshore and were accessible only by boat.

The behaviors of recreational divers were observed during regularly-scheduled dive trips run by commercial dive shops in Key Largo. Four dive shops were selected for examination of their client divers, out of 32 operating dive shops in the Key Largo area, based on 4 criteria (educational policy, dive guide service, location, and cost). In terms of the first selection criterion, dive shops were selected as either members of the Blue Star Program run by NOAA (2 dive shops selected out of 8), or non-members (2 selected out of 24). The Blue Star program was established in 2009 with the intention to reduce impacts of divers and snorkelers on coral reefs throughout the Florida Keys, through increasing public awareness. Blue Star dive shops agreed to promote reef conservation awareness through NOAA-mandated dive briefings and informative materials (pamphlets and coral identification cards) available on the dive boat and at the dive shop (http://sanctuaries.noaa.gov/bluestar/welcome.html).

The second criterion used to select dive shops was the provision of complementary dive guides during every dive. Only one dive shop in each of the two reef conservation categories (Blue Star vs. non-Blue Star) offered complementary dive guides, so these two shops were selected for examination. Selection of the other dive shop in each of the 2 conservation categories was based on location, with one shop selected in the northern region of Key Largo, and one in the southern region, because some reef sites were visited by only northern shops (3 sites) and some only by southern shops (2 sites, total of the 11 reef sites visited for diver observations, see Fig. 1). The final category for dive shop selection was cost per dive, with the two least expensive shops selected from each of the latter two groups because they were expected to attract a diverse population of recreational divers with a wide spectrum of dive experience levels. Using cost as a parameter may have created some bias in terms of the types of divers surveyed, but provided an objective, quantifiable way to select the remaining dive shops in each category. As such, the 4 shops selected for examination consisted of 2 dive shops with a dive guide (one Blue Star and one not, both in the south), and 2 shops without a dive guide (one Blue Star and one not, both in the north). All four selected shops were low cost relative to other dive shops in the Key Largo area.

2.2 Diver behaviors and impacts on corals

On each scuba dive during trips with the above four dive shops, the behaviors of 3-5 divers were observed from a distance of 3-4 m underwater (after Barker and Roberts 2004). Each diver observation was carried out either by myself and/or by one of three dive buddies who I trained and tested in diver scoring during a series of "practice" dives, to assure consistency in data collection among observers. No observations were made during the initial 5 minutes of each dive, because divers may frequently contact corals as they attempt to adjust their buoyancy at the start of each dive (Barker and Roberts 2004). Divers were selected haphazardly after they entered the water from the dive boat, and their behavior was observed for 7 min each (after Medio et al. 1997). Divers were not randomly selected while still on the boat, as done in some other studies (Medio et al. 1997, Barker and Roberts 2004), because they rapidly dispersed upon water entry and were difficult to relocate during the boat to water transition.

I quantified four types of diver contact, based on the part of the diver body that contacted live corals: hand, fin, gear (air tank, spare regulator) and other (e.g., torso, optional accessory such camera or flashlight, after Medio et al 1997, Zakai and Chadwick-Furman 2002, Barker and

Roberts 2004). I also quantified the resulting damage to live stony corals from each type of contact in four categories: sediment deposition, tissue abrasion, skeletal breakage, and no obvious damage. I recorded the period during each dive when observations began (i.e. during the first half of the dive or the second half), and whether or not the divers used accessories such as dive gloves and/or cameras (Medio et al 1997, Barker and Roberts 2004). The presence or absence of a dive guide in the water, and whether or not a dive briefing was administered before the dive, also were recorded.

During each 7-min observation period, selected divers were shadowed as they moved along the reef. Divers were not informed that they were being observed, and if they inquired about the research equipment (dive slates), I informed them that I was conducting surveys on the corals (after Barker and Roberts 2004). Divers were easily identified as belonging to a particular dive shop by the logo on their air tanks. If a diver appeared to become aware that I was observing him/her, I discarded that data set and moved onto another dive pair (this occurred with 3 of 243 total divers observed). Dive shops were also not made aware of my study in an effort to prevent altering any type of educational briefing delivered on the boat prior to water entry. This is in contrast to previous studies in the area that did alter the dive shops to their true purpose (Camp and Fraser 2012).

2.3 Variation in coral condition among sites

Within the FKNMS, 6 patch reefs (including 4 reefs visited during diver observations above, see Fig. 1) were selected that ranged from low to high diver visitation rates, as estimated from the number of mooring buoys that surrounded each reef. Mooring buoys were the primary way for commercial dive boats to secure themselves adjacent to each reef, so the number of buoys per reef indicated the relative frequencies of recreational scuba dives at each reef (pers. comm., Scott Donahue, NOAA). Little Grecian, Carysfort North, and Pickles Reef were selected as low diver visitation sites (0, 4, and 3 mooring buoys respectively), and Molasses Reef, Dry Rocks, and South Carysfort as high diver visitation sites (27, 15, and 23 mooring buoys respectively). I also selected these reef sites because they were inter-dispersed along the FKMNS (Fig. 1), were comparable in size and with the exception of Little Grecian (no mooring buoys), were visited regularly by the above dive shops.

The condition of live stony corals at each of the 6 selected reef sites was quantified using 10-12 m band transects deployed at randomly-selected locations on each reef at ~ 10 m depth. I used randomly generated numbers corresponding to degrees on a compass for the orientation of transect placement on the reef, each transect starting at 50 m distance from the mooring buoy. I then randomly sampled 4-6 1-m^2 quadrats per transect (from 20-24 possible quadrats), totaling ~ 50 quadrats examined per site (N= 309 quadrats for all 6 sites). I selected random numbers between 1 to 24 for the intervals to examine along the 12 m band transect, on either the left (quadrat #1-12) or right (#13-24). Within each quadrat, all stony coral colonies were identified as belonging to one of 12 major coral genera in 4 growth forms: branching (*Acropora*, *Porites*), encrusting *(Agaricia*, *Madracis*, *Stephanocoenia)*, massive (*Colpophyllia*, *Dichocoenia*, *Dipoloria*, *Favia*, *Montastraea*, *Siderastrea*) and foliaceous (*Meandrina*, identified using Veron 2000). These genera were selected because they included the most common stony corals in the northern Florida Keys (Goldberg 1973, Veron 2000). All other genera of corals were grouped as "others". Within each quadrat, I recorded the number and condition of all live stony coral colonies belonging to each genus, and estimated the total % live coral cover. Coral colonies that were on the quadrat edge were counted if at least 50% of the colony was within the quadrat.

Each coral colony was recorded as exhibiting 1 of 6 damage conditions: abraded tissue (tissue damage with crushed elements on the underlying skeleton), broken (fragmented or fractured skeleton, after Zakai and Chadwick-Furman 2002), sedimented (sand and other particulate matter covering at least part of the colony), tissue mortality (damage to soft tissue with no evidence of skeletal damage), diseased (distinct black band on the border between live and dead tissue, or white, moss-like tufts speckling the tissue surface, identified using Humann and Deloach 2002), or undamaged. Feeding scars made by corallivorous fishes were distinct in that a clean “scoop” of tissue and skeleton was removed by fish bites, and these scars were not considered when assessing colony condition (Hawkins and Roberts, 1992). Colonies exhibiting multiple damage conditions were categorized according to the dominant damage condition on the colony.

2.4 Statistical analyses

Statistical analyses were performed using Systat, v 13. Variation among dive shops categories in rates of diver-coral contacts and coral damage were assessed using non-parametric Kruskal-Wallis tests (Barker and Roberts 2004). Variation among reefs in coral condition also was examined with Kruskal-Wallis tests, and data on the percent cover of live corals was arcsin transformed and analyzed using one-way analysis of variance (Zakai and Chadwick-Furman 2002). All data are reported as means ± one standard deviation unless otherwise noted. For percentage data, 95% confidence intervals are shown.

3. Results

3.1 Diver behaviors and impacts on corals

Most recreational scuba divers (70.8 %, N = 240) contacted live stony corals at least once during each 7-min observation period, for an overall contact rate of 0.31 ± 0.32 contacts/min (median = 0.29 contacts/min). Thus, over the course of a typical 60-min scuba dive, each diver contacted live corals about 18 times. Divers most frequently contacted live corals with their fins (12.43 ± 1.85 contacts per 60-min dive), but also occasionally with their hands (3.04 ± 0.74) and scuba gear (dangling pressure gauges, regulators, etc., 2.32 ± 0.63). Divers also sometimes contacted live corals with other parts of their bodies (knee, elbow) or with underwater cameras or other scuba accessories such as dive knifes or compasses (0.54 ± 0.27 contacts per dive). In addition to contacting live stony corals, some divers also touched the soft sediment or sand near the reef, but these contacts were recorded only if they resulted in clouds of sediment being deposited onto nearby live corals.

In terms of impacts to stony corals from the above diver behaviors, the most frequent type of impact was sediment deposition onto the corals (9.50 ± 1.59 times per 60-min dive). Divers that directly contacted live corals sometimes abraded the coral tissue (3.18 ± 0.73) and fractured the coral skeleton (0.79 ± 0.30). About one quarter of all diver contacts with corals (including sediment deposition, 25.6% of N = 510 contacts by 240 divers) resulted in no clear damage to corals (4.68 ± 0.90 instances of no damage from observed contacts per 60-min dive).

Rates of coral contact varied significantly among divers from the 4 dive shops examined (Kruskal-Wallis Test, $P < 0.001$). Divers from both Blue Star shops contacted corals at similar rates that were significantly lower than those of divers from non-Blue Star shops, whose rates also were similar to each other (14.14 ± 2.32 vs. 22.38 ± 2.08 contacts per 60-min dive, respectively, Conover-Inman Test for Pairwise Comparisons, $P < 0.001$ for all Blue Star vs. non-Blue Star comparisons). In addition, rates of coral tissue abrasion and sediment deposition by

Blue Star divers were significantly lower than by non-Blue Star divers (Kruskal-Wallis Test, $P <$ 0.01 and $P < 0.001$, respectively; Fig. 2). Blue Star divers deposited sediment onto live corals 6.73 ± 1.52 times per 60-min dive and abraded corals 2.27 ± 0.64 times, while the rates for non-Blue Star divers were almost twice as high, at 12.38 ± 1.57 and 4.13 ± 0.81 times, respectively. Rates of diver-caused coral breakage, and of contact resulting in no obvious damage, did not differ significantly between these two groups of divers (Kruskal-Wallis Test, $P = 0.05$ and $P =$ 0.09, respectively).

None of the above measures differed significantly between divers who followed dive guides versus those who did not. Overall coral contact rates of divers with dive guides (N = 94) did not differ significantly from those of divers without guides (N = 146, 14.68 ± 1.94 vs. 18.14 ± 2.23 contacts per 60-min dive, respectively, Kruskal-Wallis Test, P=0.06). Also, none of the rates of various types of contacts or their impacts on corals differed significantly between divers with dive guides versus those without (Kruskal-Wallis Tests, $P > 0.42$ for all comparisons).

Coral contact rates of divers in the northern region of Key Largo (N = 142) did not differ significantly from those in the southern region (N = 98, 16.23 ± 2.01 contacts vs. 17.47 ± 1.66 contacts per 60-min dive, respectively, Kruskal-Wallis Test, P = 0.058). Also, rates of diver-coral contact observed during the first half of each dive (N = 123 observations during the first 30 min) did not differ significantly from those during the second half (N = 117 observations during the second 30 min, 16.58 ± 17.31 contacts vs. 20.15 ± 21.78 contacts per 60-min dive, respectively, Kruskal-Wallis Test, $P > 0.20$ for all comparisons). The number of observations made during the first half of each dive (N = 123) was larger than that during the second half of each dive (N = 117), because divers were followed during each half of each dive (total N = 240). Of the 240 divers observed, 20.4% carried an underwater camera, 20.8% wore diving gloves, and

7.1% had both a camera and gloves. Divers with a camera and/or gloves (48.3% of all divers) accounted for 57.7% of all observed diver-coral contacts, representing a significantly higher rate of coral contact than for divers without these accessories (21.6 ± 2.1 vs. 15.0 ± 1.2 contacts per 60-min dive, respectively, Kruskal-Wallis Tests, $P < 0.05$ for all comparisons). Divers with cameras contacted corals at a rate of 20.4 ± 2.4 contacts per 60-min dive, those with gloves at 22.2 ± 1.7 contacts, and those with both gloves and cameras at 24.0 ± 2.6 contacts per 60 min (Fig. 3).

3.2 Variation in coral condition among sites

The percent cover of live stony corals (both damaged and undamaged) varied significantly among the 6 examined reefs with differing numbers of mooring buoys (Figs. 1 and 4, N = 49 - 53 quadrats examined per reef x 6 reefs, ANOVA, F=86.44, $P < 0.001$). Mean live coral cover appeared low on reefs with high numbers of mooring buoys (Fig. 4), but the percent live coral cover did not vary significantly between Pickles reef (3 buoys, 22.25 ± 2.84%) and South Carysfort (23 buoys, 24.69 ± 6.48%), nor between Carysfort North (4 buoys, 32.17 ± 4.02%) and Dry Rocks (15 buoys, 25.98 ± 4.26%, Tukey multiple comparisons test, $P = 0.46$ and $P > 0.10$, respectively, $P < 0.05$ for all other comparisons, Fig. 4).

The percent of live coral colonies that exhibited some sort of damage (of all 5 types: skeletal fracture, skeletal abrasion, sedimentation, tissue mortality, and disease) also varied significantly among the 6 examined reefs (ANOVA, F = 37.46, $P < 0.001$, Fig. 4), with the reefs sorting into 2 main groups. The first group contained Little Grecian reef (0 buoys) and Pickles reef (3 buoys), which had relatively low percentages of damaged corals that did not differ significantly from each other, but had significantly lower damage rates than the other 4 reefs examined (Tukey multiple comparisons test, $P < 0.05$ for all comparisons), with the exception of Carysfort North reef (4 buoys), which differed from Little Grecian but not from Pickles Reef or any other site ($P < 0.05$ for all comparisons). The second group (Dry Rocks, South Carysfort, and Molasses Reef) also did not differ significantly from each other in their high damage rates: (15, 23, and 27 buoys respectively, $P > 0.05$ for all comparisons, Fig. 4). The lowest damage occurred on the reef with the fewest mooring buoys (0, Little Grecian) and the highest on the reef with the most mooring buoys (27, Molasses Reef). About half of all corals on the reef with no mooring buoys (Little Grecian) exhibited some form of damage, indicating possibly high background levels of coral damage in this region, likely due to factors other than diver contact (storms, pollution, bleaching, etc.). Almost all live stony corals exhibited some type of damage at the high-buoy reef (Molasses Reef, 82.11 ± 9.11%) revealing serious reef degradation at this site.

Of the five categories of coral damage surveyed, tissue mortality and sedimentation were the most common (35.65 ± 11.64 % and 29.10 ±10.92% of all examined corals, respectively), and occurred at significantly lower rates on reefs with few vs. many buoys (Kruskal-Wallis Test, $P < 0.001$, Fig. 5). Coral damage in the form of abrasion and broken skeleton were less common (13.62 ± 2.42% combined on all corals), but also occurred on a significantly higher percentage of corals at reefs with many versus few buoys (Kruskal-Wallis Test, $P < 0.05$, Fig. 4). Coral disease rates were low (4.24 ± 2.95%), and did not vary significantly between reefs with few versus many mooring buoys (Kruskal-Wallis Test, $P = 0.438$, Fig. 4).

Overall, the proportion of damaged colonies varied little among the 4 major types of coral growth forms (Fig. 5). Damage rates to branching and encrusting/foliaceous corals were similar, and most had overlapping 95% confidence intervals, except for branching colonies at Pickles Reef (0.50 ± 0.12) and Molasses Reef (0.81 ± 0.15), and encrusting/foliaceous colonies at Little

Grecian (0.43 ± 0.08) and Molasses Reef (0.62 ± 0.1). Rates of damage to massive corals varied more widely, with Little Grecian and Carysfort North grouping separately from the 3 sites with many mooring buoys (Molasses, South Carysfort, and Dry Rocks), with Carysfort North also exhibiting similar damage rates as Pickles Reef (Table 6). The 3 most abundant genera surveyed were *Agaricia*, *Porites*, and *Siderastrea* (88% of all colonies sampled, N = 1487), while the three least common were *Dichocoenia*, *Diploria*, and *Madracis* (1.2%, N = 20). *Montastraea* and *Dipoloria* colonies (N = 92), occurred infrequently but were typically large in size (1-4 m diameter) compared to most colonies of the other coral genera (< 0.5 m diameter).

4. Discussion

4.1 Overview

I show here that recreational divers cause substantial amounts of both direct (skeletal fracture, tissue abrasion) and indirect (deposition of sediment) damage to live stony corals in the Florida Keys, and that divers from shops that participate in the NOAA Blue Star program cause significantly less coral damage than do divers with other types of dive shops. I also reveal that the percent cover of live stony corals and the proportion of undamaged corals both decrease significantly with estimated recreational diving rates at 6 reef sites in the Northern Keys.

The Florida Keys attract 3 million visitors per year (Leeworthy et al. 2008), of which a substantial number participate in recreational diving. Key Largo, which is the northernmost Florida Key, only 80 km south of Miami, is the self proclaimed "dive capital of the world" (http://www.fla-keys.com/keylargo/). In the Florida Keys National Marine Sanctuary (FKNMS), which contains most of the coral reefs adjacent to Key Largo, recreational diving occurs at a frequency of 100,000-150,000 divers per year, making this one of the most intensively dived coral reef areas in the world (Leeworthy et al. 2008). The rates of coral-damaging behaviors by recreational divers that I observed at Key Largo are similar to those for divers in Australia (Rouphael and Inglis 1997), the Red Sea (Zakai and Chadwick-Furman 2002), and other areas in the Caribbean (Barker and Roberts 2004). My observations of fin kicking and deposition of sediment as the most frequent types of diver contacts with the reef, also are similar to patterns known for divers in Australia (Harriott et al. 1997), and the Red Sea (Zakai and Chadwick-Furman 2002).

These results differ from those of earlier studies in the Florida Keys, which concluded that recreational divers cause little damage to corals (Tilmant and Schmahl 1981, Talge 1992). In contrast, I document that corals reefs with high estimated levels of diver visitation in the Keys exhibit low coral cover and high damage rates, and that most divers cause some type of coral damage. Given the large increases in recreational diving rates worldwide over the past two decades (Priskin 2003, Dearden 2006), it is not surprising that current levels of recreational scuba diving in the Keys appear to be causing significant negative impacts to live corals.

4.2 Diver behaviors and impacts on corals

In addition to kicking corals with their fins, I observed divers to contact the reef with their hands, scuba gear, and occasionally their bodies (e.g. knees, elbows), and most of these contacts appeared to be inadvertent, likely due to poor diving skills and/or inexperience. Most hand contacts (68%, N = 85) occurred as divers attempted to steady themselves due to poor buoyancy, while the majority of scuba gear contacts (84% of gear contacts observed) were caused by dangling reserve regulators or dive computers, that were not appropriately tethered to the buoyancy compensator device. Even though some of these contacts resulted in no obvious coral damage, previous studies have shown that simply touching live corals can adversely affect coral health in terms of internal physiological condition (Goreau et al. 1998, Barker and Roberts 2004). Disturbances to the thin mucous layer covering corals also can increase their susceptibility to disease and algal overgrowth (Morrow et al. 2011). Such disturbances can result from even minimal contacts by divers. As such, recording of these contacts is important and should be included in future studies as potentially causing coral damage.

Previous studies also have considered many of the above types of diver-coral contacts as unintentional (Harriott et al. 1997, Medio et al. 1997, Uyarra and Cote 2007). In most cases, the divers observed here appeared to be unaware that they were harming live organisms. Therefore, reef education efforts aimed at informing divers of the major damage to corals caused by even minor physical contact could potentially substantially reduce these types of unintended contacts. Due to a current lack of government control of diver damage to corals on many reefs in Florida, reduction of the impacts of divers on these reefs relies on the actions of local dive shops, as they represent the most common interface between divers and the reefs.

These findings are similar to those of Medio et al. (1997), who demonstrated that after Australian divers were made aware that they are causing harm to living organisms, rates of unintentional contact (i.e., fin kicking, dangling gear) reduce significantly, due to dive briefings administered by local dive shops. In contrast, Barker and Roberts (2004) concluded that dive briefings do not reduce diver-coral contact rates in St. Lucia (Caribbean), but this result may have been due to inconsistencies in the briefings delivered. In the present study, the NOAA sponsored Blue Star dive shops gave a short (1-2 sentences) dive briefing prior to entering the water. All dive briefings were approximately the same length (1-2 sentences) and incorporated the same basic elements, including that divers were in a protected area and should refrain from touching or taking any corals as they are living organisms, important to the health of the reef. While brief and simple, this type of briefing reminds divers that corals are living animals and that diver-coral contact can cause harm. Unlike Medio et al. (1997), in this study I did not conduct an experiment to test whether the dive briefing per se influenced contact rates. Blue Star operators promote reef conservation awareness through a variety of strategies, including online public information, coral identification cards, and informative pamphlets. They also display the NOAA Blue Star logo on their websites, shops, and boats. This Blue Star certification thus may encourage conservation-orientated divers to use their services more than they do the services of non-Blue Star shops. These divers also may be more experienced and better able to control their buoyancy and navigation around reef structures, than are divers who are not conservation-minded. Zakai and Chadwick-Furman (2002) concluded that most diver-coral contacts in the Red Sea are caused by new or inexperienced divers. If Blue Star operators typically attract more experienced divers than do non-Blue Star operators, this could explain in part the reduced rates of reef contact that I observed by these divers. It is also possible that the dive briefing was indeed influential in mitigating coral contacts, as concluded by the only other study to examine the effect of the Blue Star program on diver behavior (Camp and Fraser 2012). In the latter study, the researchers reported frequencies of diver contacts with corals and the time in each dive that contacts occurred, but in contrast to the present study, did not record which part of the diver made contact with the reef, or quantify the resulting types of damage to corals from diver contact.

While I observed that diver-coral contact rates were significantly lower for Blue Star versus non-Blue Star divers, they did not vary with the other temporal and spatial factors that I examined. Thus, rates of diver-coral contact were uniform throughout the duration of each dive, for both types of divers, indicating that behaviors did not vary as divers were in the water longer or traveled further from the boat. Factors influencing diver behavior at the start of the dive thus appear to resonate throughout the dive, indicating that diver conservation awareness or skills acquired prior to the dive appear to have a beneficial impact on diver behavior for the entire duration of each dive. The location of the dive shop and/or reef site along the 5 km of coast examined in Key Largo also did not correlate with any examined aspect of diver behavior, indicating that the behaviors of these divers were not affected by other types of differences in characteristics or location among reefs or dive shop operations in this area.

I found that the presence of underwater dive guides did not significantly affect the rates or types of diver-coral contacts, in contrast to previous studies which observed that dive guides can alter the behavior of recreational scuba divers and significantly reduce diver-coral contacts (Barker and Roberts, 2004). A possible cause for this discrepancy may be that in previous studies studies, dive guides actively intervened when they witnessed divers contacting the reef, but in this study, they did not. The guides observed here served mainly as tour guides and swam in front of the divers, leading them to locations around the reef. Especially in cases of contact between dangling dive regulators and corals, dive guides could easily intervene and bring the issue to the attention of the diver, preventing further damage to corals during the dive.

I observed that divers using camera contacted corals at rates similar to those in Australia and the Caribbean, where camera users were responsible for high diver-coral contact rates (Medio et al. 1997, Rouphael and Inglis 2001, Barker and Roberts 2004). These findings are in contrast to those from a recent study in Key Largo which found no difference in coral contact rates between divers with cameras and those without (Camp and Fraser 2012). However, the latter study observed only 12 divers with cameras, compared to 66 divers with cameras observed in the present study, so this difference in results may be due in part to their low sample size. Some studies have suggested that divers using underwater cameras have greater diving experience (Rouphael and Inglis 2001, Barker and Roberts 2004), and thus superior buoyancy and maneuvering skills, but I observed that camera and/or glove users behaved more carelessly than did divers without these accessories. I often observed camera users to kick the reefs with their fins as they swam along trying to take pictures, leading to severe coral tissue abrasion in the form of long tread marks along the reef. Divers with cameras also crashed into and fragmented entire colonies while attempting to adjust their cameras and divers with gloves grabbed corals while they peered into crevices in search of fish and other reef inhabitants. Glove users also were more likely than were divers without to grab corals to steady themselves and help with buoyancy. I conclude that the use of underwater cameras and gloves significantly increases rates of coral damage by recreational divers in Key Largo.

4.3 Variation in coral condition among sites

Overall, on the 6 reefs examined here, as the estimated rate of recreational diving increased, the proportion of damaged corals increased and the percent live coral cover decreased. While this pattern is correlative rather than an experimental test showing a cause and effect relationship, these reefs were interspersed along the coast, and the only clear factor that differed among them was their number of buoys for dive boats. Thus, the variation in diver visitation appears to have a direct effect on the overall health of these reefs (Hodgson 1999). Examination of the two extreme reefs in terms of visitation reveals the patterns of reef damage potentially caused by divers. Molasses Reef (27 buoys) is among one of the most heavily dived reef sites in the world (pers. comm. NOAA official). A staggering 82% of coral colonies at this site exhibit at least some form of damage. Molasses Reef also has very low (12%) live coral cover, distinguishing it from all other reef sites examined here. This pattern is in stark contrast to Little Grecian (0 buoys), at which only 44% of corals are damaged and which has 38% live coral cover. Reef condition did not vary precisely with the number of mooring buoys at each site, indicating that other factors also contribute to patterns of coral condition. However, the three sites with the highest mooring buoy densities grouped separately from the two sites with the lowest densities in terms of the overall percent of damaged colonies. In terms of each type of damage observed, I also found many cases of statistical overlap between sites with different buoy densities, but the sites with more buoys generally grouped separately from those with fewer buoys.

These observed patterns of overlap of the proportion of damaged corals may be caused in part by related patterns in relative % live coral cover, because a site with less live hard coral to contact contains fewer colonies that could be damaged and vice versa. Interestingly, the sites with the two highest mooring buoy densities (South Carysfort 23 and Molasses Reef 27) also grouped together statistically in terms of the lowest % live coral cover and the highest levels of many damage types. Had levels of live hard coral cover been higher at these sites, their observed coral damage levels may have been even greater.

My observation that corals showed similar rates of damage regardless of growth form contradicts previous studies showing that branching colonies typically are the most impacted type of coral (Hawkins and Roberts 1993, Zakai and Chadwick-Furman 2002). This discrepancy may occur in part due to variation in types of branching corals among reef regions. In previous studies, branching corals typically have been medium-to-large colonies of *Acropora*, which generally form thin, long branches that project far above the reef structure. However, at my sites, live *Acropora* corals were not abundant, and the major branching form instead was *Porites*, which forms shorter colonies with fewer branches than most *Acropora* corals. Thus, the live branching corals at my sites may not have been as easily broken as the corals observed in other studies. Also in my study, coral colonies of all growth forms were ubiquitously distributed along the reef, so divers would have an equal chance of contacting corals of any growth form. Finally, I noticed many dead fragments of *Acropora* corals at my sites, which appeared to have been broken long ago. This is in agreement with previous studies showing that *Acropora* abundance has drastically been reduced in the area (Miller et al. 2002, Williams 2008). Thus, the more delicate corals (*Acropora*) had already been broken and died long before my study began, so that only sturdier, shorter branching species (i.e. *Porites*) were left on these reefs, reducing any damage differences among coral types on my reefs.

4.4 Conclusions and recommendations

Dive-based recreational tourism is an important source of business revenue in the Florida Keys. As such, the natural diversity (fishes, marine mammals) and aesthetic appearance (reef coral condition) of local dive sites represent important natural resources for the recreational diving industry. Several studies have demonstrated that dive quality and natural experience, in terms of the quality and quantity of marine life are important factors that divers consider when selecting recreational diving sites (Dixon and Sherman 1991, Kenchington 1993, and Pendleton 1994). While some studies have proposed that divers merely seek destinations with warm clear waters (Hawkins and Roberts 1994), others have found that divers care about the quality of the reefs they visit (Medio et al. 1994), and are willing to pay more to visit healthier looking reefs (Wielgus et al. 2002, Schuhmann et al. 2008). In addition, even when some of the natural features of the reef (i.e., fish diversity) do not differ much between areas with many vs. few mooring buoys, other features such as coral health may differ drastically, as was the case in this study. The reefs examined here that experienced high dive frequencies also appeared more 'kicked around', in that they contained more broken corals, fewer large-established colonies, and more patches of dead corals.

My results reveal that improved management is needed of recreational diving on Florida reefs to maintain their aesthetic natural appeal and biological characteristics. Based on these findings, I recommend the following steps: (1) Educate all dive shops to provide pre-dive briefings to recreational divers, (2) Include extra briefings for divers with cameras and gloves about potentially coral-damaging behaviors, (3) Encourage the employment of dive guides on all dives, and their active intervention when damaging behaviors by divers are observed, and (4) Promote dive shop involvement in environmental education programs such as NOAA's Blue Star program. In the current age of rapidly increasing issues in biological conservation, there is expanding public pressure to reduce activities that clearly harm the environment. The market for products harvested by sustainable means has never been greater (Vermeir and Verbeke 2006). This has also created momentum for environmental movements such as REEF (Reef Environmental Education Foundation, www.reef.org) to push for more sustainable diving practices. The NOAA Blue Star program offers a way for conservation-minded individuals to locate dive shops committed to reef conservation awareness, and also functions to inform and educate non-conservation minded divers. Increased diver demand for conservation-orientated diving operations is expected to encourage more dive shops to incorporate conservation principles into their existing operational procedures, creating a positive feedback loop that enhances both the knowledge and implementation of coral reef conservation. Reef management activities at Key Largo that incorporate the above recommendations are expected to result in a substantial reduction of diver damage to corals at this economically important tourist destination.

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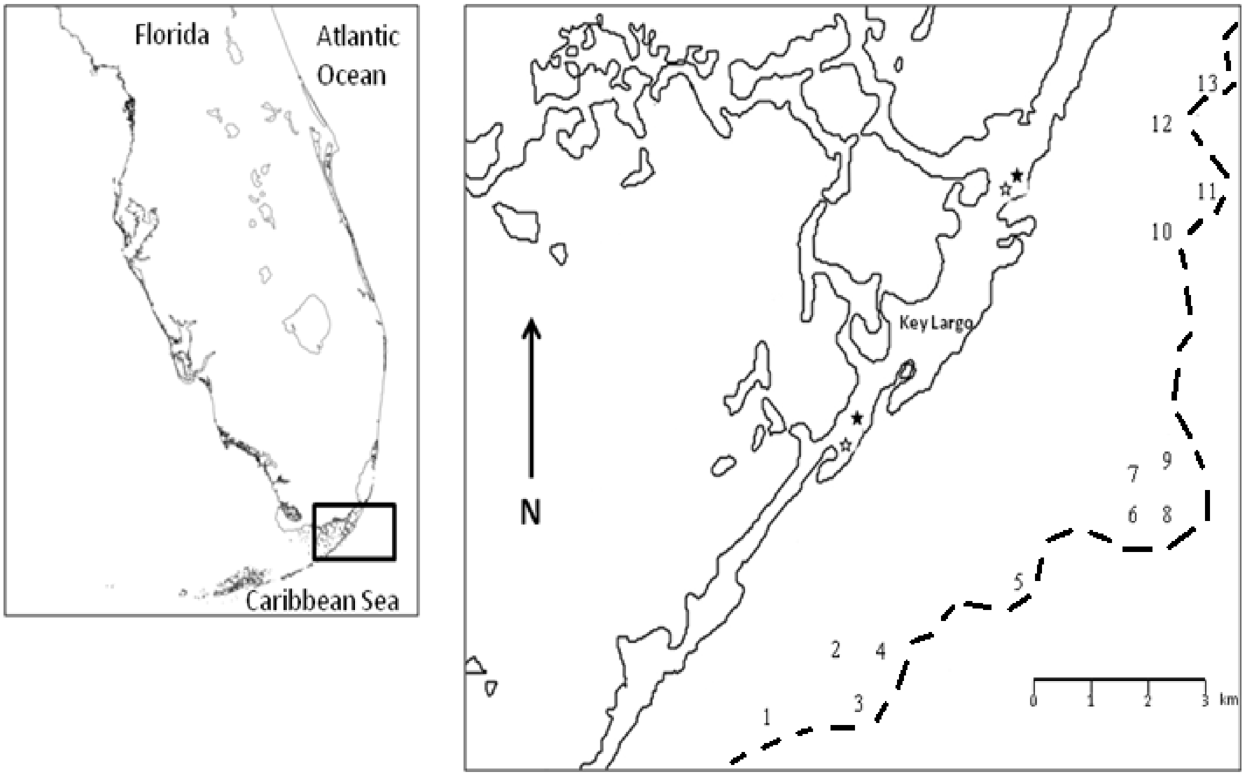

Fig. 1   Map of 4 dive shops and 13 coral reef dive sites examined at Key Largo, FL. Stars represent locations of dive shops (black = Blue Star certified, white = Non-Blue Star certified), and numbers show locations of coral reef dive sites: 1 (Pickles Reef), 2 (White Bank Dry Rocks), 3 (Molasses Reef), 4 (Sand Island), 5 (French Reef), 6 (Grecian Rocks), 7 (Dry Rocks), 8 (Little Grecian), 9 (North Dry Rocks), 10 (South Carysfort), 11 (Carysfort North), 12 (Turtle Rocks), 13 (Northeast Patch). Sites 1-7, 9, 10, 12, and 13 were visited for diver observations, and sites 1, 3, 7, 8, 10, and 11 were visited for benthic surveys of reef condition. Dotted line represents approximate 25 m depth isobaths along the edge of the Florida reef tract.

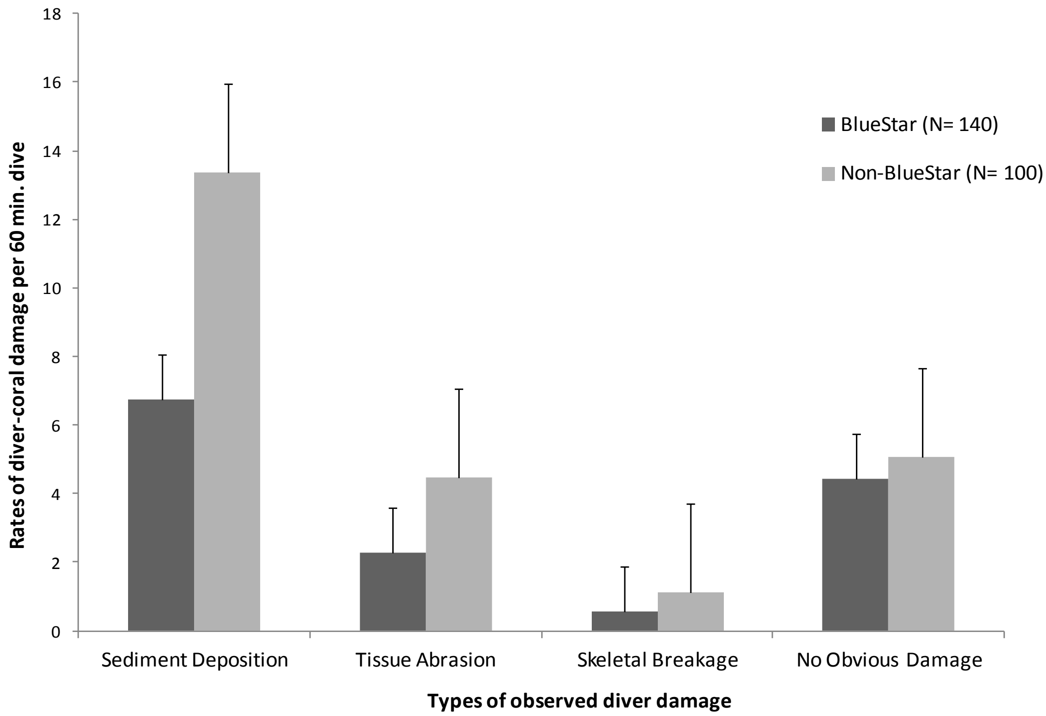

Fig. 2. Variation in rates of damage to stony corals by recreational SCUBA divers from Blue Star versus Non-Blue Star dive shops. Shown are means ± standard deviations of damage rates Sample sizes of numbers of divers observed are in parentheses.

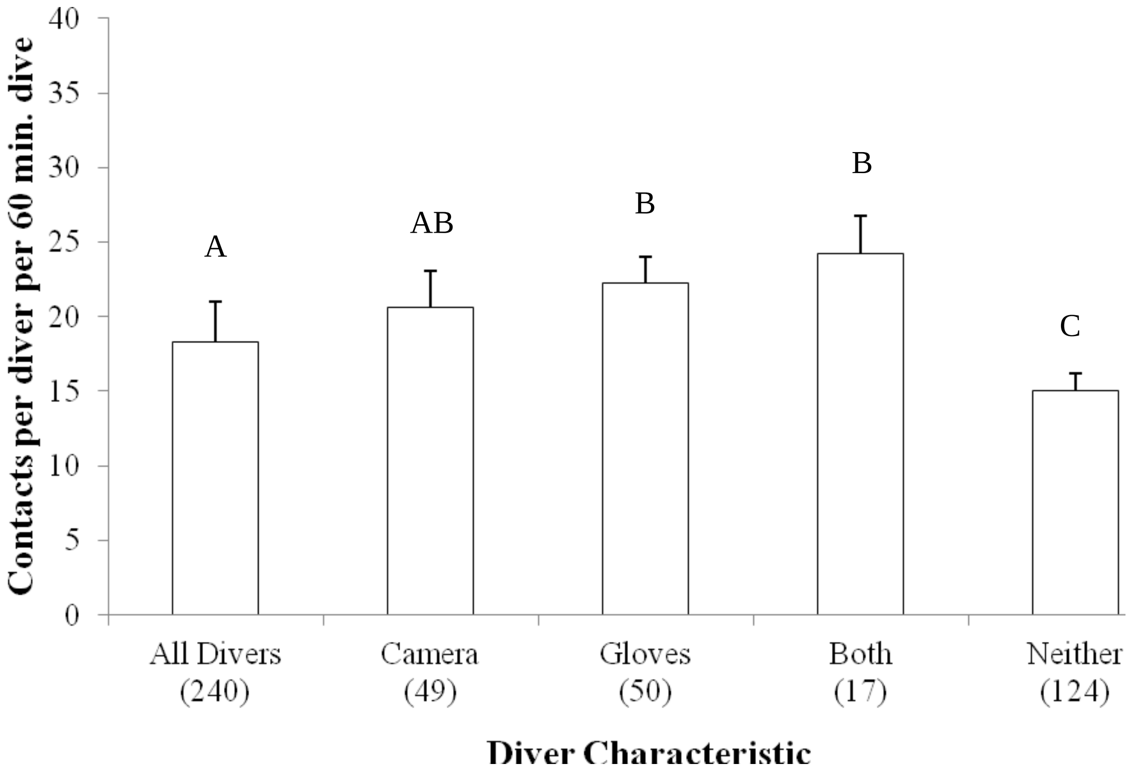

Fig. 3. Variation in coral contact rates among recreational divers with various types of underwater accessories. Shown are means ± standard deviations. Numbers in parentheses indicate number of divers observed in each category.

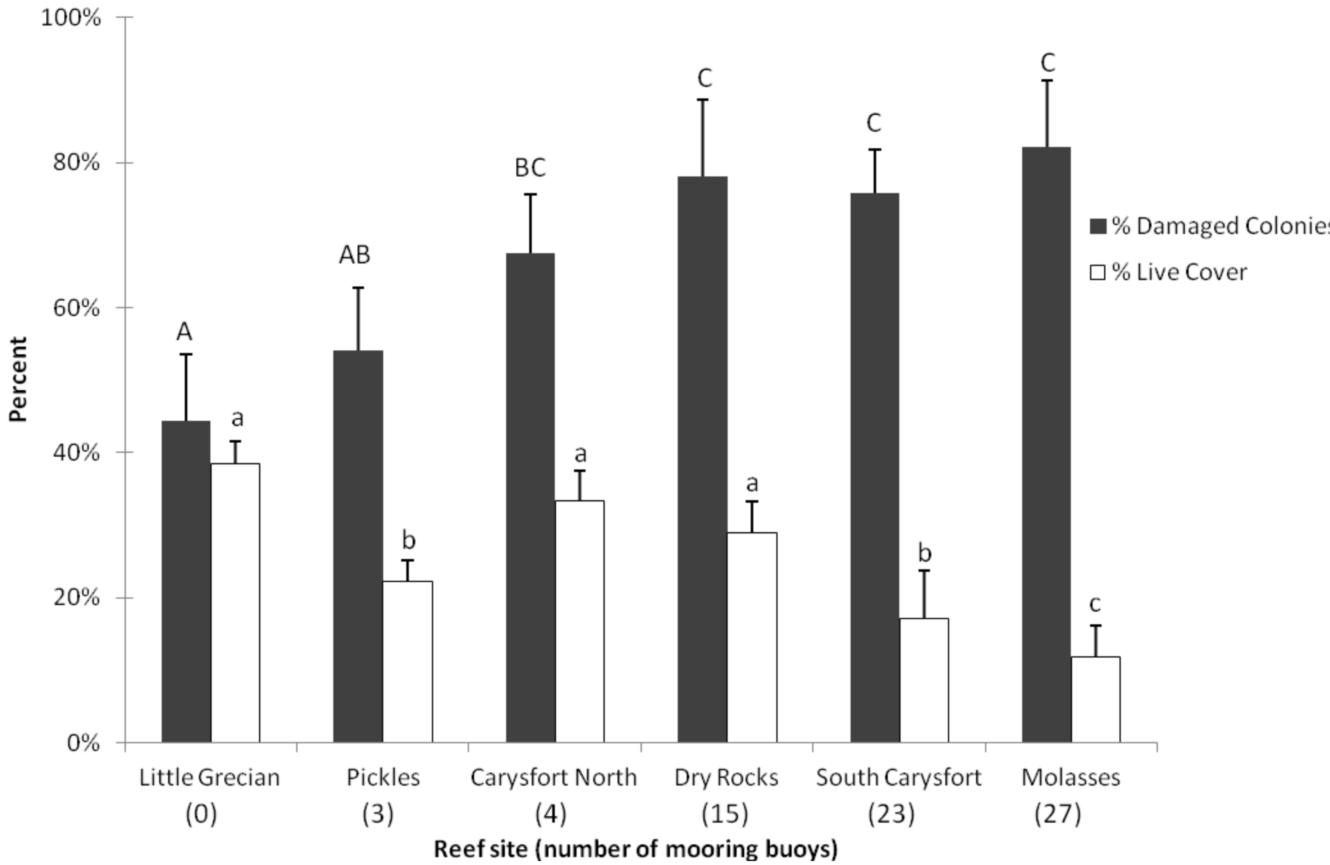

Fig. 4. Variation in the percent cover of live stony corals, and in the percent of coral colonies damaged (sediment deposited on tissue, tissue abrasion, tissue mortality, skeletal breakage, and disease) among six patch reefs at Key Largo, Florida. The patch reefs are ordered from low to high in terms of the number of mooring buoys per reef, as a proxy for the number of recreational dives per year per reef (see also map in Figure 1). Letters above the columns represent groupings based on Tukey multiple comparisons test (for % Damaged Corals, upper case) and Conover-Inman Test for Pairwise Comparison (for % Live Coral Cover, lower case). See text for details.

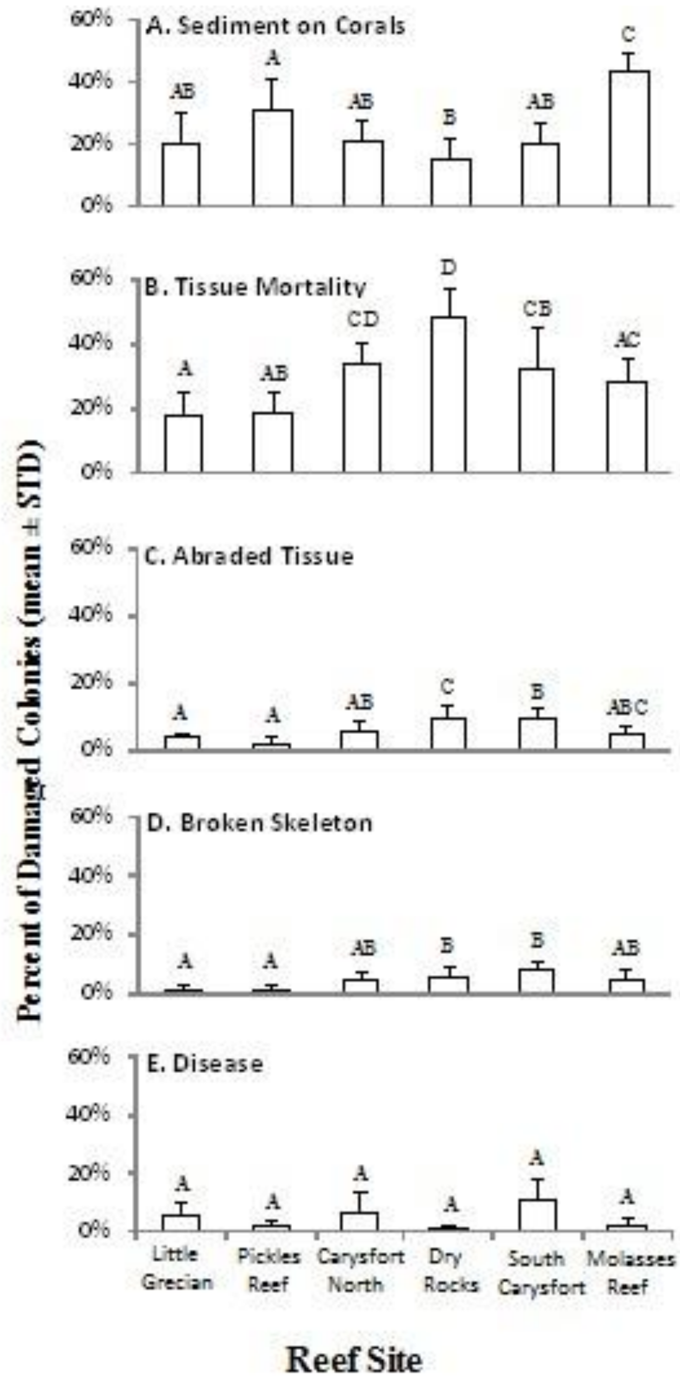

Fig. 5. Variation in rates of 5 types of damage to stony corals among 6 patch reef sites at Key Largo, Florida, ordered from low to high number of mooring buoys per reef (see also Figures 1 and 4 for site details). Letters above columns represent groupings based on Conover-Inman Tests for Pairwise Comparisons.

Damage	Growth Form	Site Little Grecian (N= 292)	Pickles Reef (N= 216)	Carysfort North (N= 185)	Dry Rocks (N= 361)	Carysfort South (N= 368)	Molasses Reef (N= 264)
Sediment on Corals	Branching	23.3 ± 7.1%	50.0 ± 9.1%	14.7 ± 5.1%	31.4 ± 10.6%	56.4 ± 12.3%	53.4 ± 11.7%
	Massive	14.1 ± 5.4%	30.7 ± 11.0%	23.4 ± 8.4%	21.5 ± 8.4%	27.5 ± 9.6%	51.9 ± 18.5%
	Other	16.8 ± 5.1%	25.6 ± 12.9%	18.7 ± 4.2%	5.4 ± 2.1%	38.4 ± 12.0%	16.7 ± 4.6%
Tissue Mortality	Branching	19.7 ± 5.1%	4.2 ± 2.1%	29.1 ± 9.2%	70.4 ± 21.2%	52.7 ± 14.3%	55.6 ± 13.7%
	Massive	14.3 ± 4.2%	24.2 ± 9.4%	36.3 ± 11.2%	46.0 ± 13.2%	32.5 ± 11.1%	27.0 ± 9.8%
	Other	18.1 ± 6.1%	17.9 ± 6.1%	35.5 ± 10.1%	26.5 ± 11.1%	31.4 ± 8.3%	47.7 ± 14.1%
Tissue Abrasion	Branching	4.0 ± 2.8%	4.4 ± 1.0%	14.4 ± 4.2%	21.1 ± 8.3%	21.0 ± 8.2%	5.7 ± 2.8%
	Massive	2.4 ± 1.1%	2.7 ± 1.3%	11.9 ± 2.3%	11.5 ± 2.3%	6.7 ± 2.2%	6.4 ± 2.7%
	Other	2.6 ± 1.0%	1.7 ± 1.0%	3.4 ± 0.9%	7.5 ± 2.3%	0%	7.6 ± 2.2%
Skeletal Breakage	Branching	2.8 ± 1.3%	4.7 ± 2.2%	11.8 ± 3.2%	8.9 ± 2.1%	15.6 ± 4.3%	10.8 ± 3.2%
	Massive	0%	1.2 ± 0.9%	5.0 ± 1.6%	6.7 ± 1.2%	8.7 ± 2.2%	3.6 ± 1.2%
	Other	0%	0%	2.4 ± 1.2%	8.6 ± 3.1%	6.0 ± 2.3%	0%
Disease	Branching	5.4 ± 2.9%	0%	0%	0%	7.4 ± 2.3%	5.4 ± 1.2%
	Massive	6.4 ± 2.1%	1.1 ± 0.7%	5.0 ± 2.3%	0%	17.7 ± 5.2%	1.7 ± 1.2%
	Other	0%	3.4 ± 1.2%	6..5 ± 2.4%	8.8 ± 2.1%	0%	0%
Total Damage	Branching	57.4 ± 12.1%	50.8 ± 12.3%	64.5 ± 13.0%	62.0 ± 14.2%	60.8 ± 13.4%	81.6 ± 15.2%
	Massive	41.2 ± 11.1%	74.8 ± 15.9%	59.4 ± 10.5%	83.7 ± 7.1%	89.4 ± 7.0%	87.8 ± 16.5%
	Other	43.4 ± 8.3%	59.6 ± 15.4%	36.7 ± 22.5%	53.6 ± 18.8%	53.1 ± 9.0%	62.7 ± 10.3%
Total Colonies Observed	Branching	60	52	45	118	141	83
	Massive	117	95	90	118	121	79
	Other	115	69	50	125	106	102

Table 1. Variation in rates of 5 major types of damage to 3 types of coral growth forms (Branching, Massive, and Other [Encrusting/ Foliaceous]) among 6 patch reef sites at Key Largo, Florida, ordered from low to high number of mooring buoys per reef. Shown are means ± standard deviations of the proportion of colonies damaged in N = 48-52 1-m² quadrats examined at each site.