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Assessment of Diver Impact During the Spiny Lobster Sport Season, Florida Keys, USA

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Assessment of Diver Impact During the Spiny Lobster Sport Season, Florida Keys, USA

by

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ABSTRACT

The Spiny Lobster (*Panulirus argus)* fishery in Florida is closed during the spawning season (March-July) except for a two-day recreational 'miniseason' for sport divers in July, several days prior to the opening of the commercial fishing season. In Monroe County, recreational fishers, who possess a valid Saltwater Fishing License with crawfish stamp, are allowed to harvest six lobsters per day, each with a minimum carapace length of 76.2 mm (3.0 inches). During these two days, approximately 50,000 people attempt to catch lobster, and the number of boats visiting the reef has been estimated to be up to 900 times higher than during the regular lobster season.

I quantified incidences of benthic damage that occurred during the August 2011 miniseason, as well as substrate type and benthos affected. Study sites at Eastern, Western, and Middle Sambos, each characterized by spur and groove reefs, represented different levels of protection within the Florida Keys National Marine Sanctuary. The Eastern Sambos is a research only area, the Western Sambos permits recreational SCUBA diving but does not allow harvest of marine resources, and the Middle Sambos allows both recreational diving and lobster harvesting. The "Impact Site", the Middle Sambos, allows lobster harvesting, and "Control Sites", The Eastern and Western Sambos, were off limits to lobster harvesting. All sites were assessed three times before and three times after the miniseason at four locations within each of the three reef areas. Research divers conducted 30-minute, random-swim surveys cataloging incidences and

magnitudes of benthic damage and counting legal-sized Spiny Lobster observed on reefs. Data were collected and analyzed using analysis of variance following the ‘Before-After, Control-Impact, Paired-Series’ (BACIPS) design.

I found an increase in the incidences of benthic damage at the Impact sites in the three surveys conducted after the miniseason, while no significant change occurred in Control sites. This suggests that detectable benthic damage associated with lobstering activity occurred during the miniseason, at least partly as a consequence of diver impacts while searching for and capturing Spiny Lobster. In addition to SCUBA gear, divers typically also bring gloves, a three-foot (92 cm) tickle stick, a hand net, a lobster gauge, and a lobster bag, all of which make buoyancy control more challenging. By actively searching for and attempting to capture Spiny Lobster, which are cryptic and maintain close proximity to the reef, lobster-seeking divers damage the benthos at higher rates than divers engaged in non-consumptive recreational activities.

INTRODUCTION

More than 100 countries have coastlines containing coral reefs, the majority of which exhibit serious declines in live coral cover (Wilkinson, 1993; Bryant *et al.*, 1998; Côté *et al.,* 2005), especially near densely populated areas (Moberg and Folke, 1999). The coral reefs of the Caribbean and Florida have changed over the past several decades, though the extent and causes of these changes remains controversial (Aronson *et al.,* 1994; Hughes, 1994; Côté *et al.,* 2005). Overall, live coral cover has decreased whereas algal cover has risen (Murdoch and Aronson, 1999). In a meta-analysis of Caribbean coral-cover data, Gardner *et al.* (2003) estimated coral cover to have decreased at examined sites from 50 to 10%, approximately 5.5% per year over the past 25 years.

Natural damages to coral reefs are typically caused by large temperature fluctuations, storm damage, and disease (Hughes *et al.*, 2003). Anthropogenic injuries have been seen due to eutrophication, pollution, sedimentation, fishing, anchoring, and diver damage (Brown and Howard, 1985). Coral reef ecosystems are widely recognized to exist in non-equilibrium conditions, consequently anthropogenic stress must be measured against this background of change (Hughes, 2002). The intensity of natural disturbances also commonly conceals effects of anthropogenic influence (Keough and Quinn, 1998).

Marine tourism has increased significantly in coral reef areas throughout the world (Davis and Tisdell, 1996; Leeworthy and Wiley, 1996). Advances in equipment have resulted in the increased popularity of SCUBA diving (Barker and Roberts, 2004), with an estimated 14 million people engaging in diving every year, many of whom seek out coral reefs (Shackley, 1998). Although diving is perceived as less damaging than extractive uses of the reef, an expansive body of literature indicates that diving is a source of reef damage (Roberts and Harriott, 1994; Prior *et al.*, 1995; Rouphael and Inglis, 1995; Harriott *et al.*, 1997; Medio *et al.*, 1997; Schleyer and Tomalin, 2000; Barker and Roberts, 2004).

Early studies (Tilmant and Schmahl, 1981; Tilmant, 1987; Talge, 1991) suggested the rates of diver use generated minor damage compared to hurricanes and natural sources. Tilmant and Schmahl (1981) conducted a three-year study in Biscayne National Park, Florida, and concluded that natural damages were more prevalent than anthropogenic damages, but found a significant correlation between reef use and physical damage, suggesting increased use would result in greater damage. More recent studies in higher use areas were able to distinguish between natural and anthropogenic damage (Rogers, 1998). Several authors have even suggested considering SCUBA diving a consumptive activity (Shivlani and Suman, 2000; Dearden *et al.*, 2007). If divers disproportionately select certain sites, the intensity of these impacts can be even more concentrated (Garrabou *et al.*, 1998; Shivlan and Suman, 2000; Lynch *et al.*, 2004; Franco *et al.*, 2009).

With an increasing awareness of anthropogenic effects on coral reefs, marine protected areas (MPAs) have become an increasingly popular choice to attempt to mitigate these potential negative effects. The MPAs, specifically no-take marine reserves, have increasingly been viewed as a practical method to protect coral reefs (Roberts and

Polunin, 1991; Pelletier *et al.,* 2005). In 1970 there were 118 MPAs worldwide, by 1980 the number had risen to 319 (Silva *et al.,* 1986), and by 1995 the number exceeded 1,300 (Kelleher *et al.,* 1995), with 400 of those containing coral reefs (Salvat and Schrimm, 2002). Boersma and Parrish (1999) analyzed more than 30 articles that documented the reasons for establishing MPAs and found that protecting marine resources and promoting or controlling tourism were among the most common.

Properly managed and monitored, MPAs have shown increased species richness, density, and average size of organisms (Bennett and Attwood, 1991; Polunin and Roberts, 1993; McClanahan *et al.,* 1999). However, these are many of the same attributes that attract SCUBA divers (Harriott *et al.,* 1997; Schaeffer *et al.,* 1999; Williams and Polunin, 2000), and several studies have found increased dive effort and impacts in response to increased protection (Schaeffer *et al.,* 1999; Shivlani and Suman, 2000; Lynch *et al.,* 2004). Recreational divers may negatively affect benthic organisms either intentionally or unintentionally (Milazzo *et al.,* 2002; Uyarra and Côté, 2007). Several researchers have examined possible SCUBA diver impacts within MPAs (Hawkins and Roberts, 1997; Rouphael and Inglis, 1997, 2001; Garrabou *et al.,* 1998; Hawkins *et al.,* 2005), sometimes identifying intense biological effects (Rouphael and Inglis, 2001; Walters and Samways, 2001; Zakai and Chadwick-Furman, 2002; Barker and Roberts, 2004). These conclusions led to SCUBA diving being considered a major form of commercial use of MPAs (Rouphael and Inglis, 2001; Franco *et al.,* 2009).

The Florida Reef Tract

The Florida Keys are an archipelago of coral and limestone islands in Monroe County and South Florida (Fig. 1), connected by a 135-mile highway (Halas and Kincaid,1993). The Florida Keys contain the only coral reef ecosystem in the continental United States and attract several million visitors annually (Shivlani and Suman, 2000). The Florida Keys National Marine Sanctuary (FKNMS), created by Congress in 1990, addressed the issue of resource protection with 26 proposed sanctuary zones, including 19 Sanctuary Preservation Areas (SPAs), three Replenishment Reserves (RRs), and four Special-use Areas (SUAs). The RRs and SPAs constitute 5.26% of the sanctuary, and the SUAs 0.02% of the sanctuary. Sanctuary regulations restrict all harvesting of marine resources within the 26 sanctuary zones. Access to special use areas (SUAs) is limited, as they are intended for research and to assess the effects of diving activities (Chiappone *et al.,* 2005).

In 1995, nearly 75% of all recreational dives in the Florida Keys occurred in the Sanctuary Preservation Areas of the FKNMS, representing 0.2% of the total area of the FKNMS, but containing many of its highly prized environments (Shivlani and Suman, 2000). Florida Keys dive operators took almost 70% of their trips and 77% of their divers to FKNMS no-take zones. In the Lower Keys, Western Sambos SPA (31 km^2) accounted for nearly 40% (12,324 divers) of trips (Shivlani and Suman, 2000).

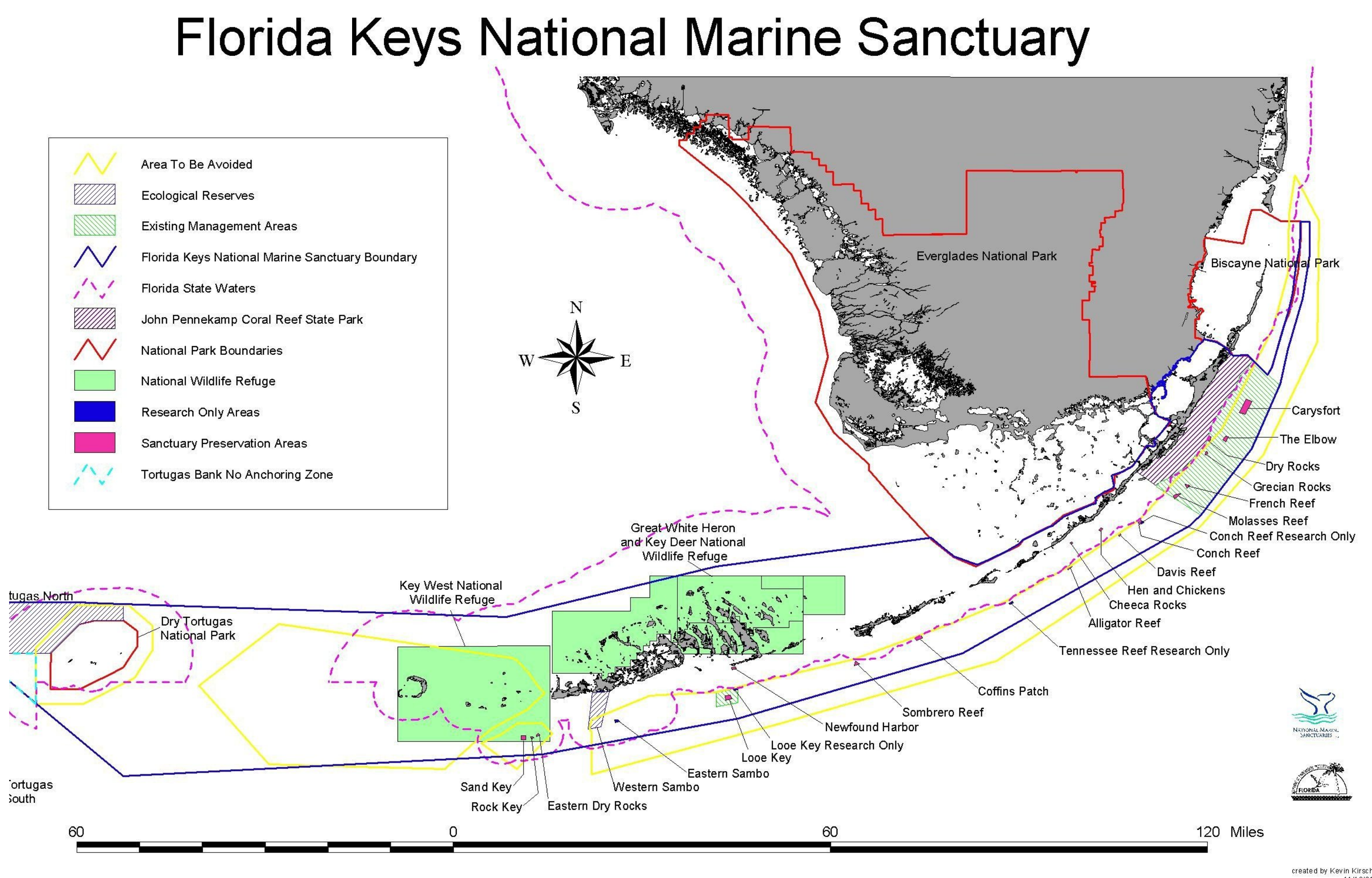

Figure 1: The Florida Keys National Marine Sanctuary (FKNMS) and the Florida Reef Tract. (NOAA)

The Spiny Lobster Sport Season

The Caribbean Spiny Lobster (*Panulirus argus)* supports fisheries from Bermuda to Brazil. Spiny Lobster (Fig. 2) enter nearshore waters from the open ocean as postlarvae and reach a legally harvestable size (in the US) of 76.2 mm (3.0 inches) carapace length (CL) approximately 30 months after settlement for females, and 23 months for males (Muller *et al.*, 1997). Season and lobster size significantly influence growth rates, with slower growth occurring among small individuals and during the winter (Forcucci *et al.*, 1994). Spiny Lobster gather in crevices during the day (Eggleston and Lipcius, 1992; Eggleston and Dahlgren, 2001), forage at night on seagrass beds and hard-bottom habitats up to 5 km away from their daytime dens (Herrnkind *et al.*, 1975; Cox *et al.*, 1997; Eggleston *et al.*, 2003), and may move as much as 200 km a year (Davis and Dodrill, 1980, 1989).

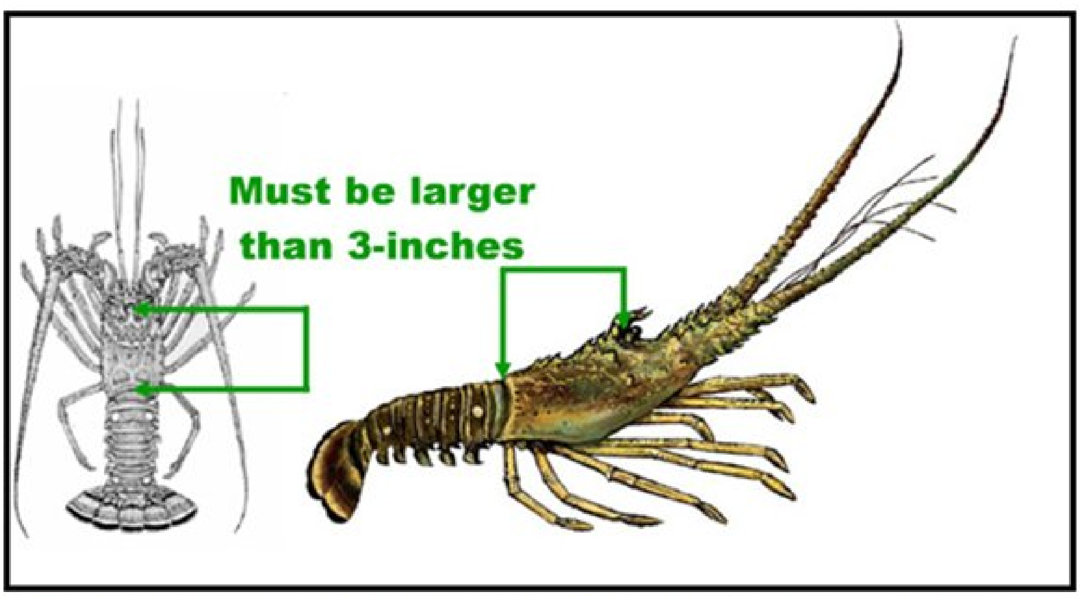

Figure 2: Illustration of measuring the carapace length (CL) of a Spiny Lobster (minimum legal size of 76.2 mm or 3.0 inches). (FWCC)

In Florida, Spiny Lobster support important commercial fisheries as well as a large recreational fishery (Sharp *et al.*, 2004). Spiny Lobster stocks are highly exploited as the fishery removes a large portion of the stock each year (Muller *et al.*, 1997); heavily relying on individuals just over the legal size limit (Beaver, 2000). Harvests from the Florida Keys account for approximately 90% of Florida's total landings (Hunt, 1994; Harper, 1995; Muller *et al.*, 1997; Cox and Hunt, 2005).

The Spiny Lobster fishery in Florida is closed during the spawning season (April-July) except for a two day recreational miniseason for sport divers at the end of July, just prior to the opening of the commercial fishing season (Eggleston and Dahlgren, 2001). During these two days, approximately 50,000 people don SCUBA or snorkeling gear in an attempt to catch lobster (Eggleston *et al.*, 2003). Every lobster fisher with a valid fishing license and crawfish stamp is allowed six lobsters per day in the Florida Keys and Biscayne National Park, and 12 lobster per day elsewhere in the state. Recreational sport divers exploit the gregarious nature of lobster by seeking out dens with high concentrations of lobster, and coercing them into hand nets with 'tickle sticks' (Eggleston *et al.*, 2003). Fishing effort is typically higher during the first day of the miniseason compared to the second, and the number of boats observed on the reef tract off the Great White Heron National Wildlife Refuge has been recorded to be approximately 900 times higher than during the regular lobster season (Eggleston *et al.*, 2003, Fig. 3). Further, the decline in lobster has been correlated with the number of recreational fishing boats observed at sites during the miniseason (Eggleston and Dahlgren, 2001).

During the miniseason, approximately 25% of the annual recreational fishing effort for Spiny Lobster can be expended (Leeworthy, 2002), and up to 255 tons of lobster are landed (Sharp *et al.,* 2005), generating an 80% reduction of the local population off Key West, Florida (Eggleston and Dahlgren, 2001; Eggleston *et al.*, 2003). Recreational sport divers account for 41% of the total lobster landings (Hunt, 2000) in August each year, and 22% of the total annual harvest, approximately 1,000 tons (Eaken, 2001). The majority of the recreational fishing effort is concentrated in the Florida Keys, comprising 60% of statewide effort, and approximately 75% of those lobster fishers live outside of Monroe County (FWCC, 2002).The socio-economic impact of the miniseason is substantial in the Florida Keys, contributing approximately $7 million to the local economy in 2001 (Leeworthy, 2002), with 95% of this economic input due to the influx of visitors (Leeworthy, 2002; Eggleston *et al.,* 2003).

The Effects of the Miniseason on Spiny Lobster

There are very few recreational fisheries in the world so intense that 80-90% of the local target population is regularly extracted in just two days each year (Eggleston *et al.,* 2003). Sport divers target dens with high numbers of Spiny Lobster and by doing this, divers disturb all lobster within that den as they attempt to capture the legal-sized individuals. Also, undersized lobster (<76.2 mm CL) are often caught and measured before release, and a large proportion of the legal lobster that avoid capture may have done so after handling and escape from divers (Parsons and Eggleston, 2007). The successful capture rate for experienced Spiny Lobster researchers has been measured as 84% on the forereef and 94% on the reef-flat (Cox *et al.,* 1997). The success rate for participants of the miniseason has yet to be measured, but the average rate would likely be lower due to inexperience. Surveys conducted in 2003 in the lower Florida Keys indicated that 27% of lobster remaining after the miniseason in coral patch reef habitats were visibly injured (e.g., missing >50% of an antenna or missing legs) and these injuries are often detrimental to lobster survival (Parsons and Eggleston, 2005).

Human disturbance and injury, representative of the levels observed during the miniseason, frequently cause lobster to abandon their usual shelters. This decreased shelter use and gregariousness after human disturbance may contribute to decreased survival when encountering a predator (Parsons and Eggleston, 2006), and injury further decreases survival by reducing conspecific attraction (Parsons and Eggleston 2005). This suggests that unintentional human disturbance may be an important component of mortality in the Florida Keys Spiny Lobster fishery (Parsons and Eggleston, 2007). Human disturbance from divers participating in the miniseason has also been documented to increase the abundance of spiny lobster in nearby marine reserves, especially if they previously contained undisturbed lobster in high numbers (Eggleston *et al.*, 2008).

Motivation for Thesis

The following questions were the motivations for this thesis:

1. Do diving activities by lobster fishers during the lobster miniseason cause detectable damage to the reef benthos?

2. Are the incidences of benthic damage significant compared to the reef's recovery ability?

The strategy used to address these questions included:

1. conducting a literature review of SCUBA diver and anchor damage typical on coral reefs,
2. designing a statistically sound survey method based on the information gained from the review,
3. testing the methods in a pilot study in the summer of 2009, and
4. conducting an intensive field study before, during, and after the miniseason in July and August 2011.

LITERATURE REVIEW

A disturbance to coral reefs is considered any change that decreases calcification or contributes to the loss of a coral reef's structure (Chabanet *et al.*, 2005). Disturbances play a crucial role in continuously shaping coral reefs and the communities they support (Connell, 1978; Brown and Howard, 1985; Done, 1992; Connell *et al.*, 1997). Disturbances can be natural (e.g., the ingestion of coral rock by parrotfish; Bruggeman *et al.*, 1994) or induced by human activities (Chabanet *et al.*, 2005). A coral reef's biodiversity may increase with intermediate levels of disturbance, but may decline if those disturbances increase in number or intensity, as the reef does not have sufficient time to recover between impacts (Connell, 1978; Hughes, 1994; Hall, 2001). Tourists eager to experience coral reefs can have significant negative effects on coral reefs by snorkeling, diving, or anchoring (Tilmant, 1987; Hawkins and Roberts, 1992; Clarke *et al.*, 1993; Jameson *et al.*, 1999; Tratalos and Austin, 2001; Zakai and Chadwick-Furman, 2002; Chabanet *et al.*, 2005).

Anchor Damage

Coral reefs are highly susceptible to vessel-based damage because they are often located in very shallow water, and their slow growth rates reduce recovery (Shivlani, 2007). Reefs large enough to accommodate a high number of vessels often suffer degradation from anchors (Walker, 2012). Anchoring is the most commonly and thoroughly studied type of vessel damage to coral reefs (Jameson *et al.,* 1999), and has been demonstrated to cause considerable and long lasting damage to coral communities (Dixon *et al.,* 1993; Tratalos and Austin, 2001). Anchors can cause damage during setting, while at anchor, and during retrieval. Corals can be broken, fragmented, or detached as the anchor is dropped to the substratum. Once set, further disturbance to the benthos is frequently associated with the anchor's chain dragging across the substratum, or entangling the reef structure (Rogers *et al.,* 1988). If an anchor becomes lodged under a coral colony, overturning can occur during the retrieval process, especially if an electric winch is used (Dinsdale and Harriott, 2004). Large anchors and chain have been observed to predominantly affect the reef's lower slope and smaller reef anchors and associated chain or rope primarily affecting the reef's crest (Dinsdale and Harriott, 2004).

As stated previously, the decline in reef-building corals is a worldwide concern. Damage inflicted by anchors on coral colonies has been recognized as problematic for several decades. For instance, in the Dry Tortugas, Davis (1977) found that up to 20% of an *Acropora cervicornis* (Staghorn Coral) zone had been destroyed by anchors. Rogers (1988) reported that 14% of vessels in the Virgin Islands National Park anchored on coral reef habitat, and over a quarter of these vessels had some impact on corals.

Increased frequencies of injured coral colonies were seen on intensely anchored sites (Rogers, 1988) similar to results reported elsewhere from coral reefs supporting high levels of human activities (Davis, 1977; Hawkins and Roberts, 1992; Schleyer and Tomalin, 2000). Jameson *et al.* (1999) compared four high-use coral reefs in the Egyptian Red Sea, and found higher levels of broken coral and rubble compared with rates of natural damage reported in the literature. Dustan and Halas (1987) recorded higher numbers of fragmented coral at Carysfort Reef (Florida Keys), which had high intensities of boating compared to nearby lower-use reefs. Lutz (2006) evaluated the condition of 315 shallow-water coral colonies from 49 sites in the upper Florida Keys and determined that nearly 60% of the sites and 80% of the coral heads showed vessel-based damage. Lutz also reported that the presence of mooring buoys did not affect the frequency of damage incidences; instead, sites near metropolitan areas and high vessel use were the most heavily impacted.

Anchors can damage corals in a variety of ways including abrasion of tissue and skeletons, death to portions of the colony, fragmentation, and detachment from the substratum (Dinsdale and Harriott, 2004). The morphology of a colony often determines its susceptibility to various types of injury (Marshall, 2000; Hall, 2001). Branching species are more prone to physical damage (Liddle and Kay, 1987), while massive and encrusting species are more vulnerable to overgrowth by algae (Hall, 2001). A coral's resistance to damage depends on the intensity and duration of the perturbation (Connell *et al.*, 1997), the geomorphology and depth of the reef zone, as well as the confounding influences of any other stresses (Connell, 1978; Hughes, 1994; Connell *et al.*, 1997). Physical destruction may not necessarily kill coral colonies, but even partial mortality may favor infestation by pathogens, and reduce the growth and reproductive potential of individuals (Hall, 2001; Chabanet *et al.*, 2005). The broken surfaces of corals will often serve as a substratum for algae and other organisms, which may infect coral tissue and further damage the colony (Bak *et al.*, 1977; Riegl and Velimirov, 1991). While anchoring impacts may have been lessened by mooring buoy installations (NOAA, 1996), anchoring on coral reefs remains a problem in developing countries (Wilkinson, 2004) where a majority of the world's coral reefs are located (Shivlani, 2007).

Anthropogenic disturbances may appear relatively minor compared to natural disturbances, such as hurricanes (Hatcher *et al.,* 1989); however, human impacts may significantly affect the recovery process of a reef, particularly if they are long-term (Connell *et al.,* 1997). Furthermore, chronic and low-level perturbations may cause more damage to the reefs in the long term than discrete and highly destructive events, because the former do not allow sufficient time for recovery (Davis, 1977; Dustan and Halas, 1987; Tilmant, 1987; Chabanet *et al.*, 2005). Even with seemingly minor perturbations such as mooring emplacements, boating, snorkeling, and diving, impacted reef communities may take several years to return to their initial structure (Underwood and Peterson, 1988). Measures at the coral colony-scale potentially provide the earliest warning of possible deterioration, while measures on the community-scale may better indicate the magnitude and ecological importance of the disturbances (Underwood and Peterson, 1988).

Diver Induced Damage

Prior studies on the impacts of recreational diving to coral reefs have consisted primarily of three types: (1) direct impact studies in which divers were followed by researchers and the frequency of contacts and extent of damage were recorded; (2) benthic surveys comparing damage in high and low use areas; (3) performing skeletal strength tests to determine a coral's resistance to physical damage, assessing tissue regeneration rates, and measuring coral-fragment survival rates (Meyer and Holland, 2008).

Direct Impact Studies

The strategy of directly observing SCUBA divers to quantify their interactions with corals and the reef was first reported by Talge (1991). At Looe Key National Marine Sanctuary, inexperienced divers were observed most often interacting with corals by inadvertently kicking them with their fins or by using corals to push themselves away from the reef. A major finding was that only a small percentage of divers were responsible for the majority of the human-coral interactions, and were mostly due to inexperienced divers with poor buoyancy control or experienced divers engaged in specific activities such as photography. Divers without gloves also had fewer interactions with the reef than those wearing gloves. These conclusions provided specific suggestions to reduce diver impact, including more emphasis on proper buoyancy control.

Subsequent studies also sought to determine patterns to identify diver impact reduction strategies, and diver training was an obvious starting point. Davis and Tisdell (1995) recognized that diving damage was mostly caused by inexperience. More experienced divers, who can better control their buoyancy, were hypothesized to have a lesser impact than inexperienced ones. Zakai and Chadwick-Furman (2002) reported exceptionally high rates of damage to corals at dive training sites, with up to 100% of all corals broken in quadrats at the most heavily used site. They found that during a typical hour-long dive at 4-8 meters of depth, each diver broke 1.7±4.9 corals.

However, not all experience-related observations have been consistent. Roberts and Harriott (1994) found that inexperienced divers (i.e., those that had completed < 100 dives) may be more likely to damage the reef, although a later study, found no such trend (Harriott *et al.*, 1997). Similarly, Rouphael and Inglis (2001) found no relationship between the level of diving experience and the number of times divers contacted the reef. Medio *et al.* (1997) demonstrated that providing divers with a 45-minute pre-dive educational briefing on the fragility of corals reduced contact with reef substrate by more than 80%. According to Barker and Roberts (2004), a one sentence addition to pre-dive briefings about not touching the reef did not reduce diver contact rates, although intervention by a dive leader was found to be very effective.

Most studies have found that a limited subset of divers were responsible for most of the damage observed, prompting research to discover other relationships between diver demographics and interactions with the reef. For example, Rouphael and Inglis (1995) reported that 17% of divers were observed to break corals during a dive, but only 4% of the divers were responsible for more than 75% of recorded damage. Some studies have noted that male divers tend to interact with the reef more frequently than female divers (Talge, 1991; Rouphael and Inglis, 2001). In contrast, Worachananant *et al.* (2008) found that male divers caused less damage per contact than females. In the Worachananant *et al.* (2008) study, observation participants were primarily female (77%), while females made up only 45% of subjects observed by Talge and 44% by Rouphael and Inglis (2001) leading to a potential sampling bias. Talge (1991) noted that a high proportion of inexperienced female divers stayed well above the reef, while inexperienced males were more likely to be negatively buoyant.

Several studies have also supported the observations of Talge (1991) who revealed the relationship between diver damage and underwater photography. Medio *et al.* (1997) noted that divers using cameras (estimated at 27%) were responsible for 72% of contacts. Rouphael and Inglis (2001) found that amateur photographers did not damage corals more frequently than divers without cameras, although underwater photographers with specialized equipment were the most damaging of all divers observed. In a study in St. Lucia, Barker and Roberts (2004) also found significantly higher reef contact by divers with cameras.

Serour (2004) found that during the Red Sea's tourist season, SCUBA divers averaged 1.3 contacts for every 10 minutes of diving, including 0.9 contacts with live coral. Photographers made up 7.2% of observed divers and were responsible for 67% of coral breakage, percentages very similar to those reported by Medio *et al.* (1997). Serour (2004) found that during months of low visitation, photographers accounted for 17% of divers and were responsible for 80% of the contacts that caused of coral breakage. Throughout the study, divers with underwater cameras were frequently observed negatively buoyant, using the reef to stabilize themselves while photographing (Serour, 2004).

In a study in Bonaire, Uyarra and Côte´ (2007), found that divers came into contact with corals more often when viewing seahorses or frogfish compared to diving outside the vicinity of these sought-after species. The authors hypothesized that the increased rate of contact was related to the cryptic nature and benthic habit of these fishes and the need for divers to come close to the substratum for observation. When divers were observing frogfish and seahorses, their contact rate increased 45 fold, and they also contacted corals for longer periods of time, both accidentally and intentionally (Uyarra and Côte´, 2007).

The importance of diver damage compared to other causes of reef impact is more challenging to determine. As with anchor damage, branching corals are more likely to be damaged by divers than other morphologies. Rouphael and Inglis (1997) reported high numbers of breaks caused by divers on reefs with high cover of branching corals. They found that up to 45% of the qualified SCUBA divers break coral colonies, but noted that the amount of damage per diver was generally small. Overall reef topography appeared to be unimportant in determining the impact rates (Rouphael and Inglis, 1997).

Meyer and Holland (2008) used handheld Global Positioning System (GPS) units to record SCUBA diver and snorkeler paths and substrate contacts over the reef. Similar to other studies, recreational impacts on coral reef habitats were found to be relatively low in number. They found that 71% of boat-based snorkelers had no contact with the substrate, compared to only 3.5% of shore-based snorkelers and SCUBA divers. Boat-based diving also had less impact per dive than shore-based diving. Overall, of the 1,340 substrate contacts recorded, only 0.7% showed obvious substrate damage and only 5% of observed contacts with live coral resulted in noticeable damage (Meyer and Holland, 2008).

In a recent study in Thailand, 93% of divers observed came into contact with the reef during a 10-minute observation period, averaging 97 contacts per hour of diving (Worachananant *et al.*, 2008). In 66% of cases divers damaged coral at least once during the 10-minute period, averaging 19 coral breakages per hour of diving (Worachananant *et* *al.*, 2008). Photographers came into contact with corals more often than other divers, causing more damage per dive, but the damage was less per contact.

Luna *et al.* (2009), in an elaborate study conducted off the coast of Spain, assessed diving experience by three different variables: total number of dives, number of years diving, and highest level of diving certificate achieved. An increased total number of dives and years of experience diving were associated with less damaging behavior. A high diving certification did not show this association and was found to less accurately reveal actual experience. This study also identified underwater photographers as making contact with the seabed more frequently than divers without cameras, as had been observed in other studies (i.e., Medio *et al.*, 1997; Rouphael and Inglis, 2001; Barker and Roberts, 2004; Serour, 2004; Uyarra and Côte´, 2007). During the observation period, 97% of divers observed had some interaction with the benthos, with an average 41.2 contacts per 10 minutes per diver. Diver's hands were found to be the part of the body that made most impacts, as has been observed in previous studies (Zakai and Chadwick-Furman, 2002; Barker and Roberts, 2004; Uyarra and Côte´, 2007). Pre-dive briefings and underwater intervention by a dive leader were also found to be effective at reducing the average impact of divers, as has been previously reported (Medio *et al.*, 1997; Barker and Roberts, 2004; Uyarra and Côte´, 2007). An observation not noted in previous studies was that divers carrying dive lights made more contacts than those without, regardless of experience (Luna *et al.*, 2009).

Benthic Surveys

Comparing among high- and low-use reef areas also requires considering possible damage by the boats used to transport divers to the reef and the damage caused by anchoring on the reef. In addition, there are impacts on water quality and sedimentation, although those issues are outside the scope of this review. The scales of impact are also different – only a small percentage of boats come in contact with a reef but when they do, the damage may be extensive. Intermediate in intensity and frequency of impact are the numerous boats that anchor on the reef. At the "thousand cuts" end of the impact scale are divers that inadvertently or deliberately contact the reef. Distinguishing among use-related impacts such as anchor damage and breakage by divers may not be a particularly important distinction, given their likely correlation. Regional differences, which may involve both taxonomic and morphological differences in coral assemblages and cultural or management-related differences in the diver populations, may also be important considerations.

In the Red Sea, Riegl and Velimirov (1991) found coral breakage to be the most common form of damage at high-use coral reef sites. They also noted damage most frequently within 10 meters of the surface, where most human interaction with the reef (snorkeling, SCUBA diving, and anchoring) takes place. In Egypt's Red Sea, Hawkins and Roberts (1992) reported finding significantly more damaged corals at heavily dived sites, recording broken coral colonies at up to 10 times higher frequencies than lightly dived reefs. More recently, Hasler and Ott (2008) also reported that reefs in the Red Sea that were subjected to intense levels of SCUBA diving showed a significantly higher number of damaged and broken corals, and lower coral cover. On the reef crest, at dived sites, over half of coral colonies were damaged and 27% were broken, with branching coral species making up over 95% of broken corals (Hasler and Ott, 2008). That study also noted that diver-related sedimentation rates decreased further from the dive site entrance, indicating poor buoyancy control was common at the beginning of observed dives, consistent with previous studies of diver-related sedimentation by Barker and Roberts (2004).

In a study in Bonaire, Dixon *et al.* (1993) determined that high dive use areas showed lower percent coral cover while species diversity was higher at lower use sites. A direct relationship between coral damage and distance from a mooring buoy was also documented, giving rise to the concept of a site's diver carrying capacity based on these findings (Dixon *et al.*, 1993). In Australia, Rouphael and Inglis (1995) concluded that a diver's lack of "environmental awareness" contributed to a greater number of impacts on the reef. Hawkins *et al.* (1999) explored the possible effects of diver-related damage to coral reef-fish communities, but ultimately did not detect any significant changes. However, a higher number of abraded corals were observed in higher use areas (Hawkins *et al.*, 1999). In Grand Cayman, Tratalos and Austin (2001) concluded that diving has had significant impacts on heavily used dive sites, as a reduction in coral cover and an increased amount of dead coral were documented as diver visitation increased.

Comparative benthic surveys have typically compared long-established dive sites with relatively undived reefs, allowing researchers to elucidate the impacts of high levels of human use on coral reefs. Most studies documented that well established dive sites had greater numbers of broken and damaged coral colonies than undived reefs nearby. However, some authors have concluded that the impact of divers may be more related to their experience and behavior than just their number (Davis and Tisdell, 1995; Rouphael and Inglis, 2001; Barker and Roberts, 2004). Dive sites also become popular for a range of biological and physical attractions not often seen on other local reefs (Tabata, 1989; Shivlani and Suman, 2000; Rouphael and Inglis, 2002). For these reasons, researchers may fail to distinguish the impacts of SCUBA diving from existing differences inherent between reefs. Nevertheless, if resource managers, dive operators, and dive educators have increased knowledge of diver behavior, more effective strategies to protect reef resources can be developed.

Manipulations of Coral Species to Physical Damage

The connection between coral-colony structure and damage resistance was first detailed by Charles Darwin in 1874, who contrasted the 'exceedingly strong honey-combed mass, which generally assumes a circular form' of *Porites* and *Millepora* colonies, which dominated the exposed edges of the Cocos-Keeling Atoll, with the 'brittle and thinly branched' species inside the protected lagoon (Darwin, 1874, page 11). Owing to their structure, *'Porites* and *Millepora* alone seem able to resist the fury of the breakers' (Darwin, 1874, page 11). Corals found in areas of greater wave energy typically a have higher skeletal strength (Chamberlain, 1975). Chamberlain (1978), using strength testing, later determined that the strength of dry, dead coral material was statistically similar to the strength of living corals. Liddle and Kay (1987) studied the effects of trampling on reef-flat coral colonies, and found massive species to be significantly more resistant to physical damage than branching species. They also found that size, morphology, porosity, and density all contribute to a coral's resistance to breakage (Liddle and Kay, 1987).

On South African reefs, Reigl and Reigl (1996) found that the likelihood of coral damage is related to its growth form, with open arborescent coral species found to be easily damaged by physical disturbances. Censuses of those fragments showed low chances of survival, especially in areas of high wave action (Riegl and Riegl, 1996). However, in Hawaii, Rodgers *et al.* (2003) found survivorship for large coral fragments to be higher than 70%, compared to 5% to 70% for small fragments. Species differences were also found to affect survival rates of fragments, with *Montipora capitata* and *Porites lobata* fragments having lower survival rates than *Porites compressa* and *Pocillopora meandrina,* two other dominant Hawaiian corals (Rodgers *et al.*, 2003).

Effects of Diving Damage

Since Ward's (1990) paper warned of the damage occurring on Florida's reefs by recreational SCUBA divers, numerous studies have investigated the impacts of diving activity and management strategies. Most have concluded that recreational diving, previously perceived as a benign activity compatible with MPA objectives, may negatively impact resources and amenity values (Dixon *et al.,* 1993; Davis and Tisdell, 1995; Harriot *et al.,* 1997; Hawkins and Roberts, 1997; Rouphael and Inglis, 1997; Hawkins *et al.*, 1999; Shivlani and Suman, 2000; Zakai and Chadwick- Furman, 2002). Divers may damage corals and other benthic organisms through contact with their hands, body, fins, or equipment (Talge, 1991, 1992; Rouphael and Inglis, 1995). Although a majority of divers come in contact with the reef during a dive, only a small percentage of them damage coral (Talge, 1991; Rouphael and Inglis, 1995; Harriott *et al.*, 1997). Most of these coral injuries are relatively minor and are unlikely to result in mortality (Talge, 1992). Healthy corals are able to regenerate tissue over relatively small injuries (Hall, 2001). However, physical damage can kill smaller colonies (Loya, 1976), and repeated injury on larger colonies can affect their capacity to recover, possibly leading to colony death (Oren *et al.*, 2001).

Coral reefs are sensitive to heavy use by divers, with documented impacts including the reduction of live coral cover and reduced abundance and diversity of corals (Hawkins and Roberts 1992, 1993; Harriot *et al.*, 1997). Degradation by SCUBA divers has been documented in coral reefs off the coasts of Australia (Rouphael and Inglis, 1995), in the Caribbean (Rogers *et al.*, 1988; Dixon *et al.*, 1993), and Florida (Tilmant and Schmahl, 1981; Tilmant, 1987). Up to 45% of qualified SCUBA divers who visit dive sites were found to unintentionally break coral colonies, though the amount of damage done per diver was generally small (Rouphael and Inglis, 1997).

Moreover, diver perception of the quality of reef resources and levels of exploitation may have economic impacts. Inglis *et al.* (1999) found that divers' experiences influence their views on crowding, and Letson *et al.* (2005) reported that diver views on reef quality may be related in part to a reef's reputation among divers. In a study conducted with live-aboard and single-day divers in Thailand, divers who witnessed diver-related damage were less willing to return for another visit (Dearden *et al.*, 2007). Similarly, in another study in Thailand, snorkelers indicated a lower satisfaction for sites that exhibited either high coral mortality or low coral diversity (Roman *et al.*, 2007).

Researchers have investigated three potential ways to minimize diving impacts to coral reefs. First, researchers have investigated diver behavior and other diver characteristics to identify groups of divers that may cause greater impacts. Secondly, research has documented the impacts of attempts to modify individual diver behavior (i.e., Medio *et al.*, 1997). Barker and Roberts (2004) found that including a short, one sentence suggestion to not touch the reef did not reduce contacts, but intervention by a dive leader did. Third, studies have investigated the relationship between the number of users and a reef's degradation to determine carrying capacity (Hawkins and Roberts, 1992; Dixon *et al.*, 1993; Harriot *et al.*, 1997; Zakai and Chadwick-Furman, 2002).

BACIPS Approach to Impact Assessment

The Before-After-Control-Impact (BACI) approach to impact assessment was originally publicized by Green (1979) and later popularized by Stewart-Oaten *et al.* (1986). Subsequently, Underwood (1991, 1992, 1994) developed the more complex "Beyond BACI" design, which demands several control sites and increased sampling times. This allows for the detection of broader forms of impacts, such as effects on variances (Osenberg *et al.*, 1994; Schmitt and Osenberg, 1996). The main hindrance to overcome in assessing environmental impacts is there is usually only one potentially impacted site (Stewart-Oaten *et al.*, 1986; Underwood, 1992). Nevertheless, Underwood (1991) stressed that several randomly chosen, similar locations should always be selected to act as controls. This allows for the recognition of natural differences among sites and for variations from the "Before" to the "After" period (Stewart-Oaten *et al.*, 1986; Osenberg *et al.*, 1996). For an impact to be reliably detected, the data should demonstrate a greater temporal interaction in the average difference between the impacted and control locations after the impact. There should also be similar temporal interaction among control locations throughout the study, allowing straightforward detection (Underwood, 1991, 1992; Stewart-Oaten *et al.*, 1992).

Multiple time designs are particularly useful when natural variation is either undetermined or pronounced (Wiens and Parker, 1995). Moreover, sampling times should be sufficiently dispersed to minimize the likelihood of significant serial correlation in the data (Stewart-Oaten *et al*., 1986; Osenberg *et al*., 1994). Serial correlation can be incorporated into BACIPS data analyses (Stewart-Oaten *et al*., 1992), but it reduces statistical power (Osenberg *et al.*, 1994). Statistical power is similarly of great importance for evaluating impacts, as it affects the likelihood of a Type II error (incorrectly concluding that a disturbance had no effect) (Eberhardt and Thomas, 1991). Statistical power can be substantially improved by increasing the number of sites sampled, the number of sampling times, or both (Underwood, 1992, Osenberg *et al.*, 1994).

The Role of the Literature Review in Design of this Study

The results of the literature review strongly supported the working hypotheses that divers heavily focused on specific activities tend to inflict more damage on reefs than recreational divers, and intense use may cause reef degradation. Another important objective of the review was to inform the design of my field research, the goal of which was to assess the impact of the intense Spiny Lobster miniseason on reefs of the Florida Keys. Underwood's (1991, 1992, 1994) BACIPS strategy provides a framework for a sound study design, within logistical and economic constraints, to offer reasonable statistical correlation between the benthic damage observed before and after the miniseason. By studying a research-only area (no damage expected), a no-take area (recreational diving damage only), and an area open to fishing (lobstering and diving damage expected), my goal was to assess the hypothesis that the effects of diving by lobster fishers on the benthos could be readily measured. This method requires near simultaneous (Paired) sampling multiple times Before and After the expected perturbation at Control and Impact sites, eliminating the problems of simpler designs, which lack spatial replication and randomization (Stewart-Oaten *et al.*, 1986; Osenburg *et al.*, 1996; Schmitt and Osenberg, 1996). Using the pilot study, it was confirmed that damage inflicted by anchors and diver contact due to the miniseason can be readily measured.

Moreover, thorough review of analysis of variance (ANOVA) and similar statistical tests (Underwood, 1992; Anderson, 2001; Connell, 2001) allowed me to avoid problems encountered in previous studies concerning the appropriate testing of the underlying assumptions of ANOVA, as well as serial correlation in the collected data (Welch, 1951; Levene, 1960; Shapiro and Wilk, 1965).

Finally, information from previous studies reporting the average algal growth rate on damaged coral and exposed hard substrata outlined the spacing of sampling dates. Algae readily colonize freshly exposed non-living substratum and dead portions of coral colonies, often within just a few days to a week, providing a means of determining whether the damage occurred recently (Fishelson, 1973; Bak *et al.*, 1977; Walker and Ormond, 1982; Littler and Littler, 1984a, 1984b, 1988; Rogers, 1988; Rouphael and

Hanafy, 2007). If surveys are spaced several weeks apart, some benthic damage that occurs between surveys would no longer be evident (Rogers, 1988).

OBJECTIVES

The primary objective of this study was to evaluate and quantify the effects of the participants of the lobster miniseason on the benthic community of a Florida Keys reef.

Secondary objectives of this study were to:

1. Using the literature review, compare behaviors or characteristics of lobster divers with divers identified as high or low impact by previous studies.
2. Design a protocol to assess recent incidences of benthic damage caused by anchoring and SCUBA diver contact.
3. Determine if 30-minute, random swims are a viable technique to survey damage to coral-reef systems.
4. Investigate whether the BACIPS study design is appropriate to evaluate very short-term perturbations.
5. Determine the suitability of using the Western Sambos research-only area as a control site for SCUBA diving damage related to the miniseason.
6. Determine if current FKNMS regulations, with respect to lobster miniseason, adequately protect reef resources within the reserves.

METHODS

A pilot study was conducted in the summer of 2009, near Fowey Rocks (off Miami, FL). Two sites open to lobstering were selected and benthic damage and the number of legal-sized lobster observed were recorded. Surveys were 45-minutes in length and were carried out once before and once after the miniseason. Both reefs experienced an increase in the incidences of benthic damage (Table 1) and a decline in the total number of legal sized lobster, although the observed change was not significant.

Table 1: The results of the 2009 miniseason pilot study off Fowey Rocks, Miami, FL.

		Before	After
Benthic Damage	Site 1	2	5
	Site 2	1	2
Legal Lobster	Site 1	2	1
	Site 2	3	2

The full-scale study was conducted in the Eastern, Western, and Middle Sambos, located approximately five miles from Key West, Florida (Fig. 3). These locations were selected for study because they are convenient to Key West SCUBA divers, represent spur and groove reefs under varying FKNMS environmental regulations, and this type of habitat has been documented to contain high densities of Spiny Lobster (Cox and Hunt, 2005). The Eastern Sambos research-only area served as a control site for both recreational diving and recreational lobster harvest (entry by researchers was allowed by permit #FKNMS-2011-077). The Western Sambos is a very popular dive location and served as a control against only recreational lobster harvest. Western Sambos was also selected as one of the controls because it has mooring buoys installed, and therefore avoids confounding the impacts of recreational divers with damage caused by boat anchoring. The Middle Sambos is located between the Eastern and Western Sambos, and allows both recreational diving and lobster harvest, and served as my expected impacted area. In each of these areas, four sites were selected based on the following criteria: (1) located in less than 10 meter water depth, (2) popular locations among divers, based on boat observations during previous years of the miniseason (with the exception of Eastern Sambos), (3) contained abundant Spiny Lobster attractive to sport season divers, based on the literature review and personal observations, (4) a minimum relief from the bottom of 1 meter, (5) within 30 minutes traveling time from Key West, and (6) at least 200 meters from any other possible study site.

The BACIPS assessment was used to evaluate the expected environmental impacts of the miniseason. This method requires near simultaneous (Paired) sampling multiple times Before and After the expected perturbation at Control and Impact sites (Underwood 1991, 1992). Each of the sites was sampled six times between July and August 2011. Each site consisted of an approximately 120 m extent of spur and groove reef with an average 8.24% living stony coral cover (Table 2) (CREMP data from Ruzicka *et al.*, 2009). Disturbances from anchors and diver contact were recorded because the damage inflicted is immediate and can be readily measured.

Benthic Damage

Since 1996, the Fish and Wildlife Research Institute (FWRI) has used the Coral Reef Evaluation and Monitoring Project (CREMP) to monitor the condition of coral reefs annually in study sites throughout the Florida Keys (i.e., Ruzicka *et al.*, 2009). The CREMP data from the Eastern Sambos Shallow and Western Sambos Shallow sites were analyzed to verify similarity among the study areas. The majority of CREMP surveys are from 1996 to 2009; two sites however, one in Eastern Sambos and one in the Western Sambos are only from 1996 to 2000 (Ruzicka *et al.*, 2009). Statistically comparing the Eastern and Western Sambos Shallow sites, percent-cover data did not show a significant difference in stony corals, Porifera, substrate, seagrass, or macroalgae (Table 2). However, significant differences were found for octocorals and Zoanthidae. Based on the data being collected, these areas were similar in the major aspects of benthic composition and therefore suitable for this study.

Table 2: Results of the analysis of CREMP benthic percent cover data between the Eastern and Western Sambos. ANOVA values are presented.

Substrate	F	DF	P Value
Stony Corals	1.8	87	0.18
Porifera	2.11	87	0.15
Substrate	2.67	87	0.11
Seagrass	0.83	87	0.36
Macroalgae	0.64	87	0.64
Octocorals	15.67	87	<0.001
Zoanthidae	64.47	87	<0.001

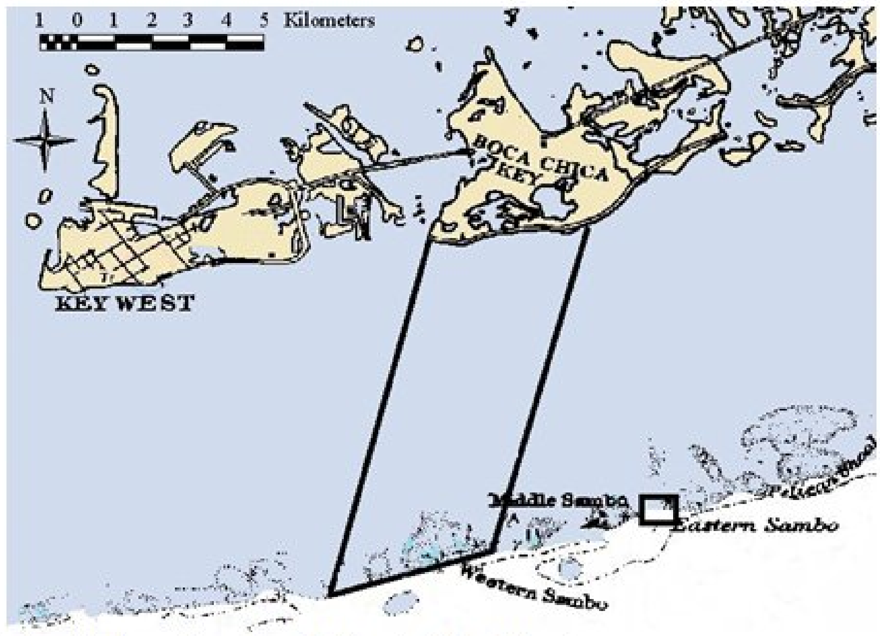

Figure 3: Study site locations in relation to Key West.

Study Design

Four sampling sites were selected within the Eastern, Western, and Middle Sambos (Appendix A). At each site four, 30-minute, random-swim (unplanned swimming route) surveys for benthic damage and legal-sized Spiny Lobster were conducted and the data recorded (Appendix B). Timed surveys were selected rather than transect surveys as the distributions of both benthic damage and legal-sized lobster have been previously observed to be quite sporadic. Additionally, timed searches allow for a greater survey area as no time is spent placing and retrieving transect tapes (Cox and Hunt, 2005). Thirty-minute surveys were chosen over shorter length surveys as damage was expected to be sparse. Coral species were also combined into broader categories, such as branching and massive, as zeroes confound BACIP analyses (Stewart-Oaten *et al.*, 1986). An attempt was made to space surveys evenly across the forereef, as coral cover and suitable lobster substrate were observed to be lower in backreef areas. Time spent inspecting damage, to determine whether it was recent, as well as measuring and recording the benthic damage's approximate area (length and width), and identifying the type of substrate affected, were not included as part of the 30-minute survey time. Time was also kept only when divers were over reasonably suitable lobster habitat (i.e., areas with solution holes and overhangs). When traversing a large area of sand or pavement, the timer was paused until the diver was back over suitable substrate.

During the miniseason the total number of boats observed at each site was recorded every hour beginning at dawn, the approved start of the miniseason. Previous studies have mentioned that an hour is the typical time it takes for lobster fishers to catch their legal limit (Eggleston *et al.,* 2003). The number of divers entering the water from each vessel was also recorded, to generate an estimate of the number of visitors to the reefs during miniseason.

This study focused on depths up to 10m, as this has been observed to be the depth range most frequently damaged (Riegl and Velimirov, 1991). Fresh coral damage was distinguished by white, exposed skeleton in the areas of skeletal breakage or tissue abrasion, which still exhibited all fine coenosteum and calicular structures, and was not yet overgrown by algae (Riegl and Velmirov, 1991). Recent damage was recorded on nine major types of substratum: non-living hard substrata (including rock and coral rubble), sand, soft corals, miscellaneous substrata, and five morphological groupings of scleratinian corals; branching, plates, foliaceous, encrusting, and massive (after Tilmant and Schmahl, 1981; English *et al.,* 1994). Digitate and sub-massive morphologies were included with the massive corals. Coral rubble was included as non-living substratum consistent with previous studies (Rouphael and Inglis, 1997; Medio *et al.,* 1997). Feeding scars made by corallivorous fish (Fig. 4) and mollusks were distinctive (Hawkins and Roberts, 1992; Schleyer and Tomalin, 2000; Rotjan and Lewis, 2008), and were not included as diver damage.

The control sites were randomly selected among sites that were similar to the potential impact site regarding coral cover, lobster presence, and ease of access to recreational lobster fishers. The survey locations were independently arranged to limit the chance of spatial autocorrelation, as recommended by Underwood (1992); surveys were spaced at intervals of no less than 200 meters. Sampling dates were sufficiently dispersed to minimize serial correlation, as recommended by Stewart-Oaten *et al.* (1986) and Osenberg *et al.* (1994).

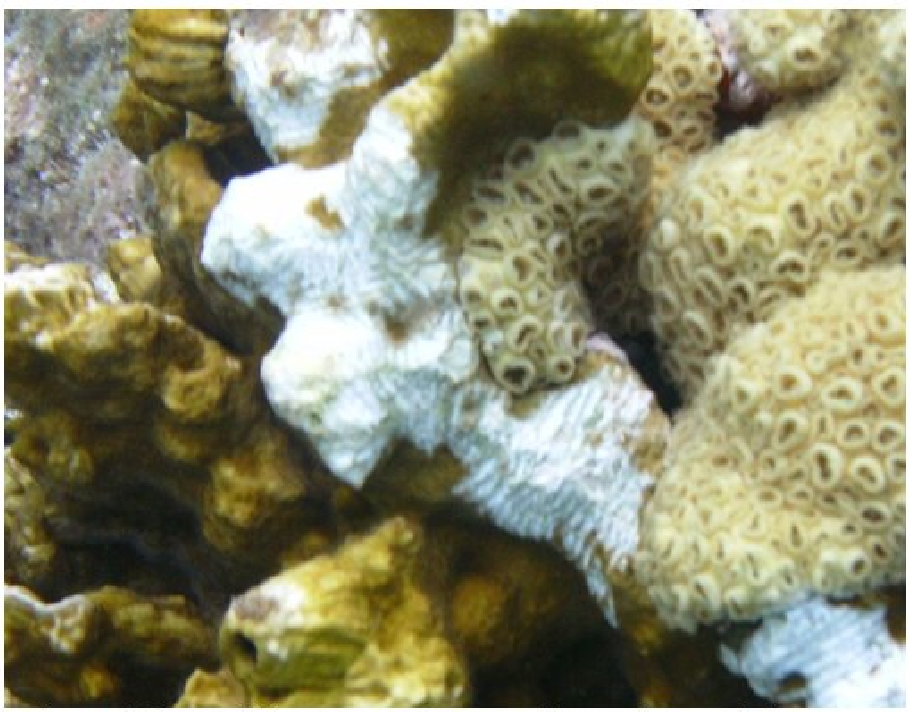

Figure 4: Photo of the parallel gouges indicative of focused parrotfish bites on a *Millepora complanata* colony. The bright white color of the exposed skeleton indicates this activity is recent.

Figure 5: Recent damage to an *Acropora cervicornis* colony. The bright white underlying skeleton and lack of algal overgrowth indicate the damage is quite recent.

Spatial and temporal patterns of benthic damage were investigated with a multifactorial sampling design. In each of the three Sambos reefs, four sites, each represented by a continuous stretch of forereef approximately 200 m long, were randomly selected. These sites were sampled six times, at randomly selected intervals centered on the miniseason, between July and August 2011. Study sites were surveyed as closely as possible (paired) and the differences were then compared to evaluate whether an impact had occurred and to estimate its magnitude, as recommended by Stewart-Oaten *et al.* (1986) and Stewart-Oaten (1996).

Statistical Analysis

The most common parametric analysis used in BACIPS studies is ANOVA, the assumptions of which are: (1) independence, each measurement has no effect on adjacent measurements, either spatially or temporally; (2) homoscedasticity, also called homogeneity of variances, the variance of the collected data should be the same within groups; (3) normality, the distributions of the residuals of the data are normal and not skewed. Testing ANOVA's assumptions are an important, though tedious, part of the analysis, as BACIPS data may not meet them (Osenberg, 1992). The advantage of ANOVA over the simpler t-tests advocated by Bernstein and Zalinski (1983) and Stewart-Oaten *et al.* (1986) is that ANOVA allows for comparison of multiple control sites and can be more decisive, especially when locations cannot be sampled simultaneously (Underwood, 1992). The independence assumption of the data (Appendix C) was examined using the Durbin-Watson test (King, 1987). A Levene's (1960) test evaluated the assumption of homogeneity of variances. The Levene test was selected rather than the more often used Bartlett test, as Bartlett's is very sensitive to non-normality in the distributions (Manly, 1998). When the result of Levene's test was significant, Welch's ANOVA is given in lieu of standard ANOVA. Welch's ANOVA uses a highly conservative estimate of the degrees of freedom to adjust for failing to meet the assumption of homogeneity of variances (Welch, 1951). The normality of the damage data was examined with the normality test of Shapiro and Wilk (1965), as it is recognized as being quite robust (Shapiro *et al.*, 1968).

RESULTS

Use Intensity as Indicated by Boat Numbers

At the Middle Sambos, the number of boats typically observed before and after the miniseason were two boats during the three to four hour sampling of study sites. On the first day of miniseason at dawn, 12 boats were observed already anchored on the reef. Boats were counted every hour and the highest number observed (18 boats) was at 10 a.m. (Table 3). Most boats departed towards shore within an hour of arriving at the reef, indicating that participants had reached their legal limit of Spiny Lobster. Vessels observed leaving the area but not in the direction of shore proceeded towards Pelican Shoals. Around noon the swells began to increase from approximately 60 cm to over 1 m, likely keeping many participants closer to shore and away from the reef. On the first day an estimated 150 boats and approximately 350 divers and snorkelers took part in the miniseason within the study area. The second day began with <1m seas, building to >1m seas, reducing the number of participants to fewer than 50 boats and approximately 100 divers. Many of the boats observed on the second day anchored at several different areas on the reef and spent more time than on the first day searching for lobster before heading to shore.

The Western Sambos had an increased number of visitors as well during the miniseason (Table 3). Fewer than ten boats were observed on the reef during surveys, many of which occurred on weekends. During the miniseason this number was typically fourteen. Lobstering is not allowed within this area and no boats were observed on the reef before dawn, although the number of vessels increased throughout the day. On the second day the number of visitors to the reef was reduced as visibility was poor.

The Eastern Sambos, which is a research-only area, contained two boats anchored and reef fishing throughout the first day. Only one vessel was observed there on the second day, which is consistent with observations throughout the study.

The number of Fish and Wildlife Commission enforcement boats observed on the reef during the miniseason was not higher than during a typical weekend. However, an increased number of FWC officers was observed at boat ramps and marinas.

Table 3: The number of vessels observed at one time within each of the Sambos on a typical day and during the miniseason.

	Regular Weekend	At Dawn, Miniseason	Peak Miniseason
Eastern Sambos (Research Only)	1	0	2
Western Sambos (No-take)	9	0	14
Middle Sambos (Lobstering allowed)	2	12	18

The assumption of independence of the data set was verified using the Durbin-Watson test statistic and averaged over each area. None of the independence tests were significant, meaning the data collected were independent. Homogeneity of variances was tested with Levene’s test and, when results were significant, the value is given and P-values derived from Welch’s ANOVA’s are used. The untransformed damage data proved normal using Wilks-Shapiro Normality test (values between 6.33E-08 and 0.0034), so no transformation of the data was necessary.

Benthic Damage Data

No significant difference in the extent of benthic damage was found when comparing all three sites to each other (Fig. 6) before the miniseason (ANOVA; $P=0.16$), and the total number of incidences of benthic damage was quite similar among sites (Table 4). Significant changes in the number of incidences of recent damage were observed after the miniseason (Welch's ANOVA; $P < 0.001$, Levene's Test $P=0.0011$), especially in the Middle Sambos, where lobstering occurred. Based on this observed change, all three study sites were then compared to themselves, before and after the miniseason (Fig. 7). ANOVA revealed no significant increase in the incidences of recent damage at the control sites (Eastern Sambos; $P=0.26$; Western Sambos; $P=0.41$). However, the impact site, the Middle Sambos, did show a significant increase in the incidences of benthic damage from before to after the miniseason (Welch's ANOVA; $P < 0.001$, Levene's Test $P=0.0015$), and the average incidences of benthic damage observed per 30-minute survey in the Middle Sambos increased noticeably (Table 5).

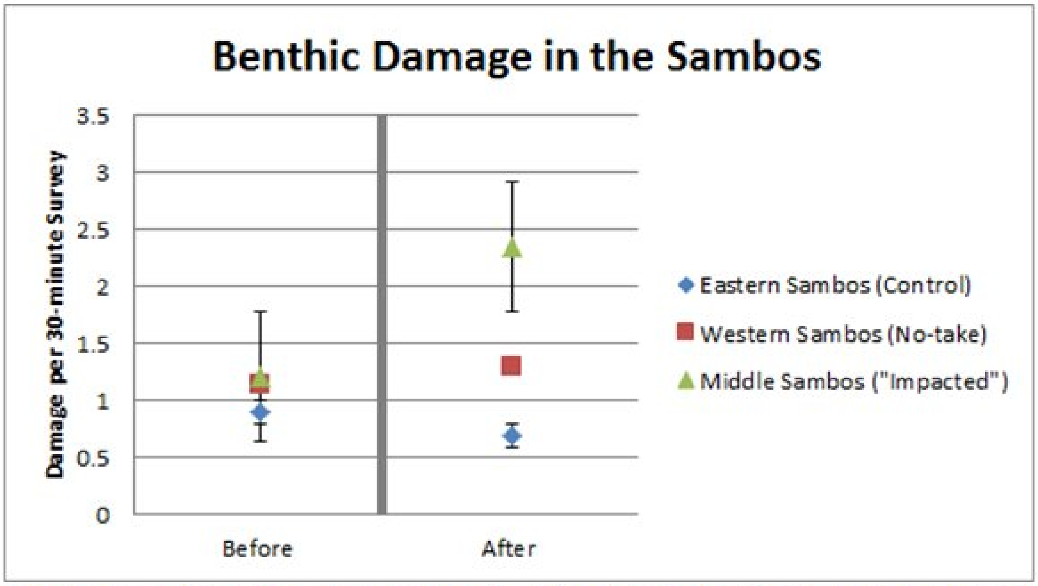

Figure 6: Average number of damage observations before or after the miniseason. The vertical bar depicts the two-day Spiny Lobster miniseason. Each point represents between 39 and 48 surveys. Note: the Middle Sambos was the only area where lobster harvesting was allowed.

Table 4: Total incidences of benthic damage observed in each study area.

	Before	After
Eastern Sambos (Research Only)	36	33
Western Sambos (No-take)	43	62
Middle Sambos (Lobstering allowed)	45	112

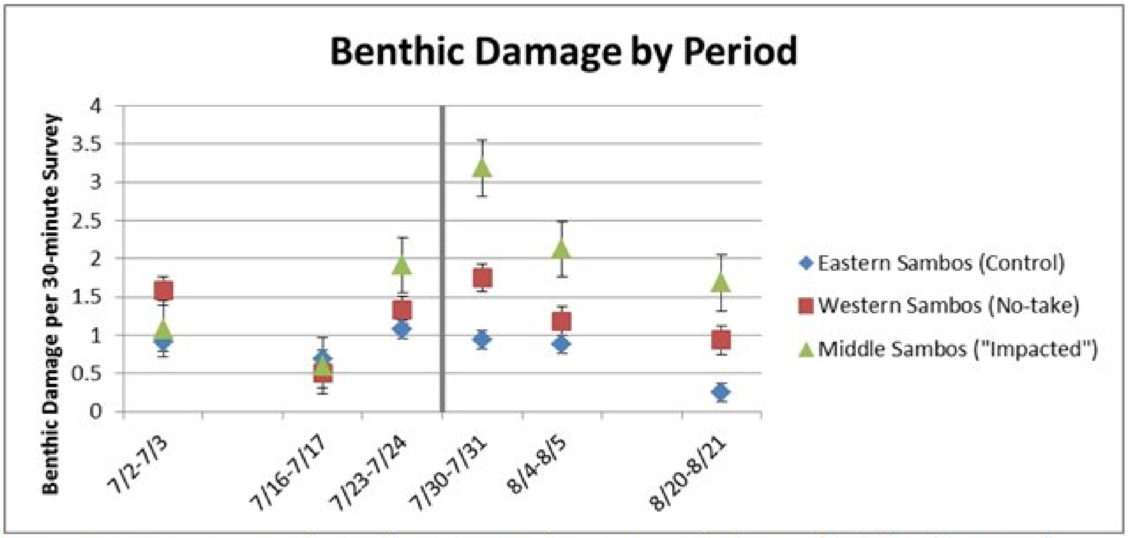

Figure 7: Average number of damage incidences recorded in each of the three study areas. Each point on the graph represents 12-16 surveys. The vertical bar in the middle of the graph depicts the two-day Spiny Lobster miniseason.

Table 5: Average incidences of benthic damage per survey on each sampling date. The miniseason took place on 7/27 and 7/28.

	7/2-7/3	7/16-7/17	7/23-7/24	7/30-7/31	8/4-8/5	8/20-8/21
Eastern Sambos (Research Only)	0.92	0.69	1.17	0.94	0.88	0.25
Western Sambos (No-take)	1.58	0.5	1.33	1.75	1.19	0.94
Middle Sambos (Lobstering allowed)	1.08	0.6	1.92	3.19	2.13	1.69

Based on the observed differences between Control and Impact areas, benthic damage data were compared by site and date. No significant differences were seen among Impact and Control areas on any sampling date before the miniseason. After the miniseason, benthic damage significantly differed between the study sites (Table 6). The sampling date of 8/4-8/5 occurred after the miniseason and before the start of the regular lobster season, and control and impact areas did not differ as markedly as when recreational lobstering was actively taking place. It is important to note that the regular lobster season opened on August 6th, possibly accounting for higher incidences of damage in the Middle Sambos on the last sampling date (8/20-8/21).

Table 6: Results of the analysis of incidences of damage at each sampling date among all the study sites. A = ANOVA values are presented, WA = Welch's ANOVA values are presented due to a significant Levene's Test value, NS = Not Significant.

	7/2-7/3	7/16-7/17	7/23-7/24	7/30-7/31	8/4-8/5	8/20-8/21
ANOVA or Welch's ANOVA	A	A	A	WA	A	WA
F	1.23	0.25	1.49	7.69	4.48	8.76
DF	35	46	35	26	47	24
P	0.305	0.782	0.24	0.002	0.017	0.001
Levene's	NS	NS	NS	0.0028	NS	0.0000125

The average size of benthic damage incidences (Table 7) was the smallest in the Eastern Sambos measuring 130 cm^2 (20.1 in^2), followed by the Middle Sambos at 133 cm^2 (20.7 in^2). The average size of benthic damage was the largest in the Western Sambos at 185 cm^2 (28.6 in^2), largely due to a single impact that measured 75cm by 90cm by 75cm high (24 in. by 30 in. by 24 in. high) (Fig. 8). With this one exceedingly large impact removed from the data set, the average fell to 116 cm^2 (18.0 in^2); as would be expected in the only area with mooring buoys and presumably fewer large impacts due to anchoring. The average depth of where impacts occurred (Table 8) was deepest in the Eastern Sambos at 3.8 meters (12.4 feet), followed by the Western Sambos at 3.5 meters (11.4 feet), and was shallowest in the Middle Sambos at 3.1 meters (10.1 feet).

Table 7: Average Size of Benthic damage observed. The average size (in cm^2) of the benthic damage observed and measured at all three study areas.

Study Area	Number	Mean	Std. error	Min.	Max.	Median	Lower 95%	Upper 95%
Eastern Sambos (Research Only)	65	129.8	56.4	1.6	903.2	38.7	18.8	240.9
Western Sambos (No-take)	154	133.5	36.7	6.5	3096.8	58.1	61.3	205.6
Middle Sambos (Lobstering allowed)	102	184.5	45.1	1.6	6967.7	51.6	95.9	273.1

Table 8: The average area and depth of benthic damage within each of the Sambos. *Denotes average with the largest single impact excluded.

	Average size (cm^2)	Average depth (meters)
Eastern Sambos (Research Only)	130	3.8
Western Sambos (No-take)	116*	3.5
Middle Sambos (Lobstering allowed)	133	3.1

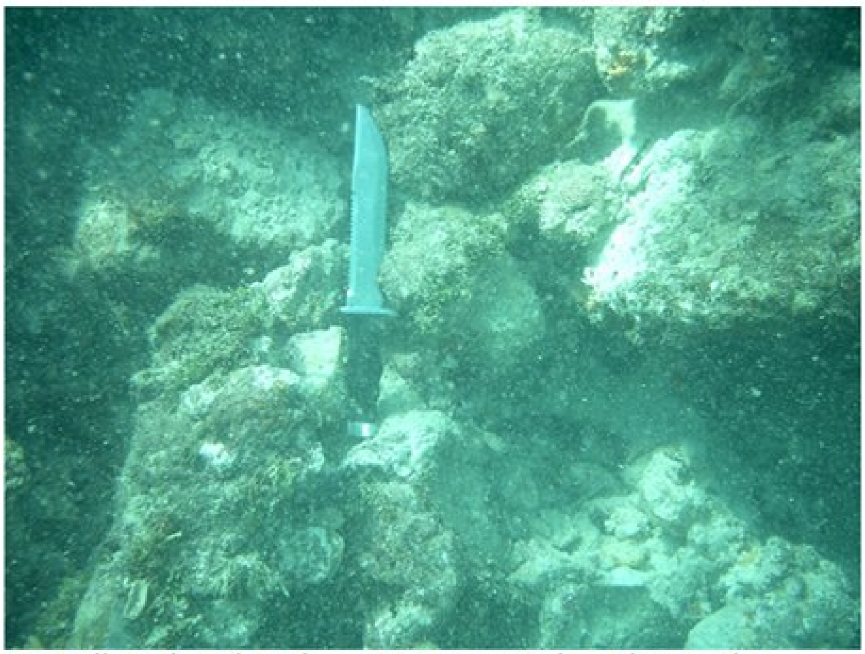

Figure 8: Collapsed coral overhang in Western Sambos. This was the largest benthic impact recorded during the study and measured approximately 0.75m by 0.9m and was 0.75m high. The dive knife pictured measures 32 cm in total length.

The most frequently observed damage incidences involved stony corals, when all types of coral were combined. Stony corals made up on average 8.2% of benthic cover within the Sambos (based on analysis of CREMP data by Ruzicka *et al.*, 2009), but 44% of benthic damage observed throughout the study; while the number of damaged corals observed from all sites combined nearly doubled (71 observed before the miniseason to 141 after), they more than tripled in the Middle Sambos (20 observed in the before period to 72 observed after). The most commonly damaged coral was *Millepora alcicornis* (Branching Fire Coral) followed by *Millepora complanata* (Blade Fire Coral), together making up 46% of all incidences of coral damage, or 20% of all observed damage. During the study, damage to four colonies of *Acropora cervicornis* (Staghorn Coral) and one colony of *Acropora palmata* (Elkhorn Coral) were observed, all of which consisted of detached branches, comprising 1% of observed damage. Damage to non-living substrate, comprised 24% of all recorded damage.

Spiny Lobster Survey Data

Some of the observed differences in *P. argus* density among surveys are possibly due to the differences in the visual survey proficiency of volunteer divers and the cryptic nature of Spiny Lobster. Lobster data (Appendix D) were normally distributed, except for the 7/16-7/17 surveys, but in some cases variances were not equal. By comparing the Before and After period split by each of the Sambos, it was possible to see how each responded to the miniseason. In the Before period, the three Sambos were significantly different (Table 9). The Eastern Sambos had the lowest average number of lobster observed, followed by Middle Sambos, and Western Sambos contained the highest number. Post miniseason, the sites were no longer significantly different from one another (Table 9). The Eastern Sambos exhibited a slight increase in the average number of lobster observed, while the Western and Middle Sambos both decreased. Although not quantified, the number of injured lobster observed on the reef post-miniseason increased noticeably, as has been recorded in previous studies (Parsons and Eggleston, 2005).

Table 9: ANOVA results of Spiny Lobster presence. A = ANOVA values are presented, WA = Welch's ANOVA values are presented due to a significant Levene's Test value, NS = Not Significant.

	Before	After
Eastern Sambos (Research Only)	3.55	4
Western Sambos (No-take)	6.53	6.23
Middle Sambos (Lobstering allowed)	5.23	5.21
ANOVA or Welch's ANOVA	W	A
F	5.81	2.14
DF	75	141
P	0.0045	0.12
Levene's	0.051	NS

DISCUSSION

The primary objective of this study was to evaluate and quantify the effects of the lobster miniseason on the benthic community of a Florida Keys reef. My approach was to test the hypothesis that a reef subjected to intensive lobster fishing for two days would display significantly more damage than nearby control reefs. Indeed, before the miniseason, the incidences of benthic damage recorded were not significantly different among the three reefs assessed, despite their strikingly different visitation levels (analogous to Tilmant and Schmahl, 1981; Muthiga and McClanahan, 1997). However, after the miniseason, the differences in incidences of benthic damage were significant. In control reefs, the Eastern and Western Sambos, the miniseason did not result in a significant increase in the incidences of damage seen, but the Middle Sambos experienced a significant increase in damage; experiencing a more than doubling of incidences of benthic damage, of all types, recorded (Table 4). The Middle Sambos also differed most markedly from control areas on the sampling dates of active lobstering.

Lobster Divers as High-impact Users

The first of my secondary goals was to compare behaviors and characteristics of lobster divers with recreational divers previously identified as high or low impact. Numerous studies have demonstrated that certain recreational diving activities are more likely to damage benthos, including corals, than other activities. Diver's behaviors, experience, type of activities, as well as certain physical attributes of a coral reef, all influence the potential for impact (Medio *et al.*, 1997; Rouphael and Inglis, 2001; Barker and Roberts, 2004). The two groups of SCUBA divers commonly recognized as most likely to interact with the substrata include a) inexperienced divers, especially those with poor buoyancy control; and b) divers engaged in specific activities that bring them close to the reef, focus their attention, affect their buoyancy control, or any combination of those behaviors (Talge, 1991; Medio *et al.*, 1997; Rouphael and Inglis, 2001; Walters and Samways, 2001; Barker and Roberts, 2004; Serour, 2004; Uyarra and Côte´, 2007; Luna *et al.*, 2009). Examples of the latter group include divers carrying and using photographic equipment, divers observing small or cryptic species within the reef (and often photographing them), and divers carrying dive lights. These recreational specialists often display goal-oriented behaviors at odds with other values (Ditton *et al.*, 1992). Consequently, taking advantage of an opportunity to photograph a unique or cryptic species may override environmental concerns, and lead them to underestimate the environmental impact of their actions (Rouphael and Inglis, 2001; Uyarra and Côte´, 2007).

In the hands of less skilled divers, cameras or other equipment greatly impair a diver's buoyancy control, which is the cause of most damaging incidents (Rouphael and Inglis, 2001). Luna *et al.* (2009) concluded that carrying anything causes divers to have more interactions with the environment than divers who do not carry something.

Thus, lobster fishers are characteristic of the second group of "high-impact" divers. While lobstering, divers wear gloves and carry a hand net, a three-foot (92 cm) fiberglass "tickle stick", and lobster bag, all of which make buoyancy management much more difficult. The goal-oriented focus of capturing Spiny Lobster also distracts divers from avoiding contact with the benthos. Lobster-seeking divers spend much of their time in close proximity to the reef, and inadvertently or deliberately contacting benthic organisms while attempting to capture lobster.

Was the Research Design Suitable?

Previous studies of diver impact and of the impact of specific diving activities have confronted and often identified specific challenges related to the design of statistically rigorous studies. Most studies that have applied the BACIPS design have been several years long, consistent with the perturbation being measured such as a nuclear power station installation. With only a two-day long disturbance and minimal natural variability in the data being collected, a BACIPS study spanning only 8 weeks proved to be sufficient to detect significant changes in the incidences of benthic damage at the Middle Sambos.

Using timed, random-swim surveys, more replicate surveys could be conducted in each study site on each sampling date, which improved the precision of the estimates of incidences of benthic damage. Random-swim surveys of 30-minute lengths were effective for the type of data being collected and most divers were able to conduct three surveys with a single 80 cubic foot dive tank.

Thus, the results of this study demonstrated that the experimental design could produce statistically rigorous results. Although directly observing participants of the sport season may have provided a more accurate estimate of impacts, according to Eaken (2001), most (> 90%) of lobster fishers utilize their own boats and most of the rest use rental boats. Thus, impartial, direct observation of diver behavior as performed in previous studies (Medio *et al.,* 1997; Rouphael and Inglis, 2001; Zakai and Chadwick-Furman, 2002; Barker and Roberts, 2004; Serour, 2004; Uyarra and Côte´, 2007; Meyer and Holland, 2008; Worachananant *et al.,* 2008; Luna *et al.,* 2009) was not possible to meet the objectives of this study.

Managing the Impact of Diving in the Florida Keys

Relatively few of the observed injuries to coral colonies, which were assumed to be the consequence of damage by divers, likely resulted in coral mortality, based on the results of the study by Talge (1992). Moreover, the very low numbers of incidences of damage on control reefs, averaging just over one observation per 30 minute survey, indicate that direct damage by recreational diver activities is usually minimal (Table 5). The inclement weather during the 2011 lobster miniseason undoubtedly limited diving activity, especially on the second day, thereby perhaps limiting the impact of divers. However, the argument can be made that the surge associated with one meter waves, which would be experienced by divers working in 3-5 m depths, might increase the direct impacts of the divers on the reef. In addition, the low visibility associated with the wind and wave activity would require divers to work more closely to the substrata than if conditions were more ideal.

Coral colonies detached by anchor damage can suffer high mortality, especially if they settle in the sand between spurs (personal observations). Unlike, Tilmant and Schmahl’s (1981) previous observation of stony corals averaging a lower percent damage than their proportional occurrence (less than 5% of the total damage in Biscayne National Park), these corals made up on average 8.2% of benthic cover within the Sambos (based on Ruzicka *et al.,* 2009 data), but 44% of benthic damage observed throughout the study, with *Millepora* comprising 46% of all coral damage. This may have been partly a consequence of the relative abundance of *Millepora* in the Sambos reefs; this genus has previously been reported as more susceptible to damage relative to its abundance in the community (Tilmant and Schmahl, 1981). However, *Millepora* is a somewhat “weedy” species (Sullivan and Chiappone, 1993) and is not a member of the major reef-building Order Scleractinia.

The largest single impact observed during the study occurred in the Western Sambos (Fig. 8), ironically the only site with mooring buoys installed. This impact appeared to have been the result of an anchor becoming caught under a reef overhang and the vessel using an electric winch to pull it loose, causing the overhang to partially collapse. Thus, as other studies have noted, the presence of mooring buoys may not necessarily eliminate anchor damage (Lutz, 2006).

Evidence of poaching was observed sporadically throughout the study, especially at the Western Sambos. Several recently lost pieces of lobstering and fishing equipment were found during the study including a tickle stick, hand net (both before miniseason), and a speargun band with metal wishbone attached. Divers were also observed exiting the water with a large speargun on one occasion. Eastern Sambos sporadically had vessels anchored and actively fishing, and one recently lost reef anchor was recovered. Although no divers or snorkelers were ever observed there, some recent diving-related debris was also found.

Coral reefs like those found in the Sambos, which are easily accessible, are more likely to exhibit symptoms of diver or anchor damage, as the intensity of use of a reef is closely related to its distance from the nearest boat ramp (Kenchington, 1984). While significant differences in the levels of damage were documented between protected reefs and those open to lobstering, the incidences of damage to the benthos were relatively low. Moreover, despite the Western Sambos popularity among commercial dive boats and recreational SCUBA divers, the incidences of damage on this heavily used reef averaged 1.2 observations per 30 minute survey (Table 5), and were not significantly different from that recorded on the research-only Eastern Sambos reef. Thus, the 60% increase of incidences of benthic damage on the Middle Sambos reef just after the lobster miniseason is more striking in percentage than in absolute magnitude (Appendix A).

The results of this study mirrored Hawkins and Roberts (1992), who concluded the differences in damage may be relatively unimportant biologically 'but aesthetically were striking' (page 178). Other studies have concluded approximately 4% coral damage is considered to be ecologically sustainable (Hawkins and Roberts, 1997; Hawkins *et al.*, 1999; Jameson *et al.*, 1999). Nevertheless, coral reproduction and growth may decrease after damage (Allison, 1996) and the extent of coral damage may ultimately affect the survival of the colony (Meesters *et al.*, 1996).

Because the reefs of Florida Keys have a high ease of access and SCUBA diving is essential to the local economy, managers must work with the constraints that some level of damage to benthic organisms will occur. The question is how much damage is tolerable; as it has been observed that the conversion to 'diver-damaged' reefs can occur very quickly (Hawkins and Roberts, 1992). Clearly, direct damage by divers participating in the miniseason is presently not a significant cause of reef decline in the Florida reef tract, and other factors associated with human activities are much more likely to be detrimental. At present, the extremely low abundances of branching corals (Callahan *et al.,* 2006), which are the most prone to breakage by divers, may limit recorded incidences of damage by diver activities.

CONCLUSIONS

1. Observed benthic damage in the Middle Sambos more than doubled, comparing the period before to after the miniseason.
2. The peak number of vessels observed on the study reef during the miniseason was approximately nine times higher than that observed during a regular weekend.
3. There was no significant difference in recorded incidences of benthic damage comparing study reef and control reef sites before the miniseason.
4. Recorded incidences of damage on the study reef immediately after the miniseason were approximately double what were recorded at sites before the miniseason and at the control sites after the miniseason.
5. The BACIPS design, using 30-minute surveys, proved effective at detecting significant differences between reefs subjected to intense lobster diving and the control reefs.
6. No significant differences were detected in the size or depth of damage area between study and control reefs.
7. Inclement weather during the 2011 Spiny Lobster miniseason likely reduced the number of boats and divers on the study reef, thereby possibly limiting the impact of the miniseason.
8. Given the nine-fold increase in boats observed on the study reef during the miniseason, the more than doubling increase in the incidences of damage indicates that only a small percentage of lobster fishers are impacting the reef.

9. The study design did not provide data that could determine whether the diver damage during the miniseason was ecologically significant.

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APPENDICES

Appendix A: Study Site Coordinates

East Site 1	24° 29’ .610 N	81° 39’ .599 W
East Site 2	24° 29’ .559 N	81° 39’ .675 W
East Site 3	24° 29’ .481 N	81° 39’ .848 W
East Site 4	24° 29’ .510 N	81° 39’ .906 W
Wes Site 1	24° 28’ .856 N	81° 42’ .708 W
Wes Site 2	24° 28’ .970 N	81° 42’ .227 W
Wes Site 3	24° 28’ .939 N	81° 42’ .364 W
Wes Site 4	24° 28’ .917 N	81° 42’ .539 W
Mid Site 1	24° 29’ .376 N	81° 40’ .344 W
Mid Site 2	24° 29’ .344 N	81° 40’ .408 W
Mid Site 3	24° 29’ .328 N	81° 40’ .471 W
Mid Site 4	24° 29’ .292 N	81° 40’ .582 W

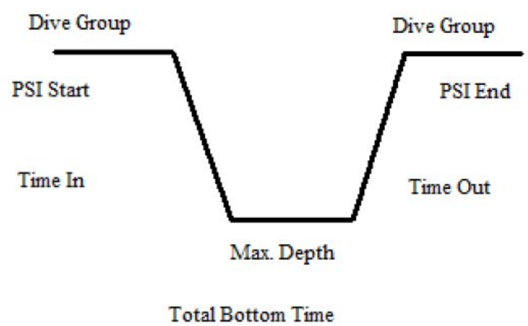
Study Site Coordinates representing the four sites chosen within each of the three areas. The northernmost site in each of the Sambos is listed as Site 1.

Appendix B: Example of Benthic Damage Data Sheet

Name ___________________________________ Date _____________

Site _______________________________________ Depth_____________

Damage	Substratum type	Length (inches)	Width (inches)	Depth	Picture number
1					
2					
3					
4					
5					
6					
7					
8					
9					
10					



P. argus seen: _____________

Guide to Substratum Type.
Scleratinian corals

- Branching (BR)
- Foliaceous, (FO)
- Massive (MA)
- Plates (PL)
- Encrusting (EN)

- Non-living hard substrata (HS) Including coral rubble, and pavement
- Soft corals, gorgonians, sponges (SC)
- Sand, (SA)
- Miscellaneous substrata (MI) (e.g. algae, mushroom corals, clams, etc.)
- Marine Debris (MD) Will include lost diving and snorkeling gear as well as anchors.

Note that it must not have algal or other marine growth on it.

This is an example of the benthic damage survey sheet used throughout the study. Volunteer divers were given underwater digital cameras and small (7.5cm by 10 cm by 1.6mm thick) magnetic numbered sheets (numbered 1 through 10) to allow recorded damage in each survey to be reviewed thoroughly. Magnetic sheets were selected because they are heavy enough to resist the wave action on the reef and are not easily lost underwater.

Appendix C: Benthic Damage Survey Data

	7/2-7/3	7/16-7/17	7/23-7/24	7/30-7/31	8/4-8/5	8/20-8/21
East Site 1, Survey 1	1	1	2	2	2	0
East Site 1, Survey 2	4	0	1	1	0	0
East Site 1, Survey 3	1	1	1	0	0	0
East Site 1, Survey 4	0	1	X	1	0	1
East Site 2, Survey 1	2	0	3	2	2	0
East Site 2, Survey 2	1	2	1	2	1	0
East Site 2, Survey 3	0	0	2	0	2	0
East Site 2, Survey 4	1	2	X	2	1	0
East Site 3, Survey 1	0	1	1	0	1	0
East Site 3, Survey 2	0	0	1	0	0	1
East Site 3, Survey 3	1	0	1	0	0	0
East Site 3, Survey 4	0	2	X	1	1	1
East Site 4, Survey 1	X	1	1	2	0	1
East Site 4, Survey 2	X	0	0	0	0	0
East Site 4, Survey 3	X	0	0	1	3	0
East Site 4, Survey 4	X	0	X	1	1	0
West Site 1, Survey 1	3	1	0	2	1	3
West Site 1, Survey 2	3	0	2	4	0	0
West Site 1, Survey 3	0	0	2	3	3	0
West Site 1, Survey 4	1	0	X	1	2	2
West Site 2, Survey 1	1	0	4	1	2	2
West Site 2, Survey 2	0	2	0	0	0	0
West Site 2, Survey 3	1	0	2	2	0	0
West Site 2, Survey 4	3	0	X	0	0	1
West Site 3, Survey 1	3	1	1	3	1	0
West Site 3, Survey 2	2	2	2	4	0	2
West Site 3, Survey 3	0	0	1	0	1	2
West Site 3, Survey 4	2	1	X	3	1	0
West Site 4, Survey 1	X	1	2	1	0	0
West Site 4, Survey 2	X	0	0	2	4	1
West Site 4, Survey 3	X	0	0	1	4	1
West Site 4, Survey 4	X	0	X	1	0	1
Mid Site 1, Survey 1	1	0	3	6	3	3
Mid Site 1, Survey 2	2	0	0	7	1	0
Mid Site 1, Survey 3	0	0	2	2	2	1
Mid Site 1, Survey 4	1	X	X	1	1	1
Mid Site 2, Survey 1	X	1	2	4	3	3
Mid Site 2, Survey 2	X	0	0	3	1	3
Mid Site 2, Survey 3	X	0	1	0	1	1

Mid Site 2, Survey 4	X	0	X	1	2	4
Mid Site 3, Survey 1	2	2	4	7	3	2
Mid Site 3, Survey 2	1	2	3	3	1	0
Mid Site 3, Survey 3	2	0	1	4	5	0
Mid Site 3, Survey 4	0	1	X	0	3	2
Mid Site 4, Survey 1	2	1	2	4	3	4
Mid Site 4, Survey 2	1	1	3	4	0	0
Mid Site 4, Survey 3	1	0	2	4	3	0
Mid Site 4, Survey 4	0	1	X	1	2	3

This table shows the data collected during the benthic damage surveys. Each number is the result of a 30-minute survey conducted by one diver. An X represents a survey that was not completed (due to inclement weather or too few divers). Damage to Soft Corals was excluded from the dataset as occurrence was highly inconsistent and was markedly increased with rough sea conditions and increased wave action.

Appendix D: Lobster Observation Data

	7/2-7/3	7/16-7/17	7/23-7/24	7/30-7/31	8/4-8/5	8/20-8/21
East Site 1, Survey 1	4	1	3	1	2	2
East Site 1, Survey 2	13	12	2	7	1	6
East Site 1, Survey 3	4	10	1	3	1	9
East Site 1, Survey 4	4	6	X	0	0	0
East Site 2, Survey 1	0	3	2	11	3	6
East Site 2, Survey 2	2	10	1	0	4	0
East Site 2, Survey 3	1	1	0	7	1	6
East Site 2, Survey 4	0	7	X	3	1	0
East Site 3, Survey 1	2	3	4	5	2	0
East Site 3, Survey 2	1	6	3	2	0	6
East Site 3, Survey 3	4	4	1	4	3	35
East Site 3, Survey 4	1	3	X	7	1	1
East Site 4, Survey 1	X	2	3	2	2	1
East Site 4, Survey 2	X	4	5	10	2	1
East Site 4, Survey 3	X	4	3	2	7	20
East Site 4, Survey 4	X	2	X	1	4	0
West Site 1, Survey 1	4	6	4	16	6	2
West Site 1, Survey 2	2	9	5	18	4	7
West Site 1, Survey 3	0	6	4	3	2	22
West Site 1, Survey 4	3	7	X	17	4	2
West Site 2, Survey 1	4	8	2	4	2	8
West Site 2, Survey 2	5	9	7	5	6	4
West Site 2, Survey 3	2	4	4	1	10	3
West Site 2, Survey 4	1	4	X	2	4	1
West Site 3, Survey 1	5	15	4	6	3	2
West Site 3, Survey 2	3	16	10	6	2	2
West Site 3, Survey 3	2	6	3	4	1	0
West Site 3, Survey 4	1	8	X	3	1	6
West Site 4, Survey 1	X	9	10	19	7	10
West Site 4, Survey 2	X	12	26	8	2	8
West Site 4, Survey 3	X	8	14	19	9	15
West Site 4, Survey 4	X	9	X	10	3	0
Mid Site 1, Survey 1	1	X	8	5	4	3
Mid Site 1, Survey 2	5	13	4	3	9	1
Mid Site 1, Survey 3	1	8	4	0	7	7
Mid Site 1, Survey 4	1	5	X	0	5	2
Mid Site 2, Survey 1	3	2	9	3	10	7
Mid Site 2, Survey 2	7	12	12	0	6	3
Mid Site 2, Survey 3	0	6	3	2	20	8

Mid Site 2, Survey 4	0	4	X	2	14	2
Mid Site 3, Survey 1	1	2	4	5	2	1
Mid Site 3, Survey 2	4	8	8	8	9	6
Mid Site 3, Survey 3	4	6	4	2	10	5
Mid Site 3, Survey 4	0	8	X	8	5	2
Mid Site 4, Survey 1	X	3	12	7	1	1
Mid Site 4, Survey 2	X	7	6	5	10	5
Mid Site 4, Survey 3	X	8	3	6	9	11
Mid Site 4, Survey 4	X	8	X	1	8	0

This table shows the numbers of lobster observed during the benthic damage surveys. Each number is the result of a 30-minute survey conducted by one diver. An X represents a survey that was not completed (due to inclement weather or too few divers). Volunteers were not specifically searching for Spiny Lobster, only recording those observed during the benthic damage survey.