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The Effect of "Casitas" on Lobster Biology and Fishery Sustainability in the Bahamas

Lester George Gittens
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THE EFFECT OF "CASITAS" ON LOBSTER BIOLOGY AND FISHERY SUSTAINABILITY IN THE BAHAMAS

by

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A Dissertation Submitted to the Faculty of
Old Dominion University in Partial Fulfillment of the
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ABSTRACT

THE EFFECT OF "CASITAS" ON LOBSTER BIOLOGY AND FISHERY SUSTAINABILITY IN THE BAHAMAS

Lester George Gittens
Old Dominion University, 2017
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“Casitas” (artificial table-like structures) are a commercial fishing gear used to harvest Caribbean spiny lobster (Panulirus argus) in the Caribbean and in The Bahamas, where lobster is the most valuable fishery. Yet, casitas are largely unregulated in The Bahamas and they may threaten fishery sustainability through alteration of lobster growth, disease, or mortality rates and due to insufficient information concerning their number and location. Focusing on the lobster fishery in The Bahamas, my objectives were to: (1) investigate the mortality, growth, and susceptibility to disease of lobsters collected in casitas compared to wooden traps and those living in natural shelters; (2) evaluate lobster size-structure and the bycatch of undersized lobster and other taxa from casitas compared to traps; and (3) ascertain whether remote sensing can be used to estimate the distribution and density of casitas.

Using tethering and time-lapse photography, I compared predation on lobsters at casitas and natural locations. I also compared the growth, frequency of injury, and prevalence of disease of lobsters collected from casitas and traps, as well as lobster size structure and the bycatch of other taxa and of undersized lobsters from traps and casitas. Fishermen use hooks to remove lobsters from casitas, so I also tested if hooking increased the mortality of under-sized lobsters released after capture. In situ measurements of the reflectance of casitas relative to background habitats along with the analysis of remote sensing images were used to explore the feasibility of using remote sensing to detect and census casitas in Bahamian waters.

Predation of lobsters at casitas and natural shelters did not differ, nor was the survival of sub-legal sized lobsters affected when impaled by a hook during removal from casitas. A larger size range of lobsters occurred under casitas than were caught in traps. However, lobsters in traps were in poorer health than those in casitas. Casitas were detectable using remote sensing but refinement of detection methods is needed before the technology can be employed by fishery
managers. All in all, there is no evidence that casitas have a negative ecological effect on lobsters in The Bahamas.
Copyright, 2017, by Lester George Gittens and Mark J. Butler IV, All Rights Reserved.
This dissertation is dedicated to my family, especially my wife Deline, my sons Matthew and Nathan and my daughter, Leah. They motivate me to always do my best and to always pursue good. This also includes my parents Edward and Stephanie Gittens, my mother in law, Jacqueline Dyce and those members of my family that were present when I started my PhD studies but were no longer here to witness its completion, including my uncle Solomon Hem-Lee, my grandmother Neresa Hem-Lee and my father-in-law Stafford Dyce.
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CHAPTER I
INTRODUCTION

Biological, social, and economic factors all affect fishery sustainability (Martinet et al. 2007) and the reciprocal is also true: namely, these factors are in turn affected by fishery sustainability. Indeed, to achieve large-scale conservation and economic security, the sustainability of marine fisheries is necessary (Charles 1997). A variety of indicators are used to measure sustainability including ecological, biological, and socioeconomic indicators (Caddy 2004, Boyd and Charles 2006). These may include more specific gauges such as the adequacy of institutional management resources, level of community cooperation, biodiversity indices, harvest value, safety of fishers, conflict resolution, protein availability, fish waste amounts and direct effects of fishing gear on ecosystems just to name a few (Boyd and Charles 2006). Although a variety of indicators may be used, sustainability is difficult to measure (Dahl 2012) because measurement is influenced by vague targets (Cochrane 2000) and complicated interdependencies between the aforementioned indicators. In this dissertation, I examine the effects of the expanding use of a specific type of fishing gear (casitas) on the biology and sustainability of the fishery for Caribbean spiny lobster (*Panulirus argus* Latrielle, 1804) in The Bahamas.

*The Fishery for *P. argus*

My investigation focuses on understanding certain biological aspects of the *P. argus* fishery that are necessary for management. However, economic considerations highlight the critical need for the proper management of this resource in The Bahamas.

Landings of *P. argus* in the Caribbean were recently valued at $800 million (FAO/WECAFC 2007) and the commercial fishery employed an estimated 33,000 persons (Chávez 2009). The fishery for *P. argus* is also the most valuable in The Bahamas, where in 2015 over 2,000 metric tons of lobster were landed valued at $55 million. This was mostly lobster “tails”, as the cephalothorax is normally removed and discarded at sea. This roughly equates to 6,500 metric tons when converted to whole weight for standardized comparison to other countries. In 2015, lobster landings represented approximately 64% of landings by weight
and 84% of the value of commercial landings in The Bahamas according to Government of The Bahamas records. Among fourteen countries in the Western Central Atlantic region that land appreciable amounts of spiny lobster, The Bahamas caught the equivalent of 20% (6100 MT whole weight) of the 30,600MT of lobster landed in the Caribbean during 2013 (FAO 2015). Lobster from The Bahamas is exported to the United States of America, Europe, Canada, China, Japan, Germany and parts of the Caribbean (Government of The Bahamas records). This is probably an underestimate of the dominance of The Bahamian lobster fishery in the Caribbean, because in some countries reporting includes other species of spiny lobster [e.g., *Panulirus guttatus* (Latreille, 1804) and *Panulirus laevicauda*, (Latrielle, 1817)] and occasionally lobsters from other families (slipper or "shovel-nosed" lobsters; Scillaridae). Worldwide, approximately one billion people have an inadequate protein intake (Wu et al. 2014), so lobsters also help to meet global protein requirements although lobster has been considered a luxury food item for decades (Graham 1968, Bowen 1980).

The Bahamas has a number of fisheries management measures in place aimed at ensuring the sustainability of the fishery. These include minimum size limits for the cephalothorax and “tail”, a four month closed season during the period of peak spawning, prohibition of the taking (and possession) of egg-bearing females, a “bag limit” for foreign recreational fishers, a harvest control rule based on catch rates, licensing requirements for hookah diving (surface supplied air using an air compressor), wooden lathe trap size limitations, exclusion of non-residents from commercial fishing, and the banning of SCUBA for fishing. Of these management measures, regulations that control the fishing gear that can be utilized have been in place since the mid-1980s. At that time the predominant fishing methods included wooden lathe traps or free diving / skin diving. In recent decades, fishermen in the Bahamas and other countries in the Caribbean (e.g., Cuba, Mexico) have increasingly turned to the use of "casitas" or "condos" as fishing gear, rather than traps. By the mid-1990s, the use of casitas became the most widely used fishing method in The Bahamas. However, there are no measures in place to record or limit the use of casitas, although all of the existing management measures (e.g., closed seasons, size restrictions, etc.) also apply to persons using casitas.

Casitas are artificial structures deployed on the seafloor that mimic the natural crevice shelters used by lobsters. They are rectangular and tend to be approximately 1.8m long x 1.2m wide with a top made of corrugated metal and sides composed of 1.8m lengths of 5cm x 15cm
lumber that raise the top approximately 15cm off the seafloor (Figure 2, Chapter II). Casita designs vary and may include car hoods for tops and sections of pine trees for the legs. Fishers have long capitalized on lobster social behavior to increase their catchability in traps and now in casitas (FAO 2007). Lobsters are collected from casitas by divers or snorkelers; the casitas are thus left on the seafloor during both the open and closed fishing seasons and are not pulled out onto the vessel during sampling as are traps. Of particular concern about the use of casitas in The Bahamas is that their number is unknown, but estimated to be hundreds of thousands to over 1 million (Doerr 2014). Also of concern are the possible effects that casitas may have on key aspects of lobster biology, including growth, disease, and mortality.

_Lobster Life History and Ecology_

Although the economic importance of _P. argus_ fisheries is evident, less easily quantified is the ecological importance of the species, a common scenario among large decapods (Boudreau and Worm 2012). As a predator, it is suspected that a reduction in the numbers of _P. argus_ could result in an increase in the numbers of the species upon which it preys (Davis and Dodrill 1980). Based on stomach content analysis, this primarily includes hermit crabs, brachyuran crabs and gastropods (Briones-Fourzán et al. 2003). A reduction in prey where lobsters are common has indeed been observed in temperate ecosystems where the effects of lobster predation on dominant prey species (e.g., mussels, sea urchins) has cascading effects on communities (Breen and Mann 1976, Robles 1987, Shears and Babcock 2002, Blamey et al. 2010, Matassa 2010). A dramatic example of this was observed in waters off California where fishing reduced the population of _Panulirus interruptus_ (Randall, 1840), which led to a proliferation of urchins that resulted in a boom-bust cycle of urchin disease and similar cycles of excessive grazing on kelp (Tegner and Levin 1983, Lafferty 2004). Only one example of strong trophic effects of lobster predation has been reported in a tropical ecosystem (Butler and Kintzing 2016), where the mere odor of _P. guttatus_ (but not _P. argus_) reduces grazing by _Diadema antillarum_ (Philippi, 1845), an important herbivore on Caribbean reefs (Kintzing and Butler 2014). As adults, _P. argus_ grow large enough to defend themselves against most predators, but as juveniles they are a prey item for a variety of fishes and invertebrates (Smith and Herrnkind 1992).

Although diet varies by location, _P. argus_ is a secondary consumer (Behringer and Butler 2006b) and, given its abundance, may play an important role in the transfer of energy and
biomass between trophic levels. If so, then significant declines in the abundance of *P. argus* due to over-fishing could alter food webs, including indirect effects on organisms that neither feed on nor are eaten by *P. argus*. A few "back of the envelope" calculations provide an estimate of the amount of carbon that is removed from the marine environment on the Great Bahama Bank due to the lobster fishery. Net primary production on the Great Bahama Bank (approximately 96,000 km² surface area) has been estimated to be $2 \times 10^{13}$ g carbon per year (Dierssen et al. 2003). Lobster landings in the amount of 2,000 metric tons of tails per year equate to approximately $2.7 \times 10^8$ g C yr$^{-1}$, assuming that lobsters are comprised of approximately 50% carbon (dry weight) like other animals (Houghton 2003) and a wet weight to dry weight conversion of 26.8% (Schram and von Vaupel Klein 1999). This represents the equivalent of the removal of approximately 0.01% of the NPP of Great Bahama Bank, which appears small. Lobster carapaces were excluded from calculations because they are typically discarded at sea. Nonetheless, decomposers and scavengers would also be impacted by over 4,000 metric tons of lobster carapaces discarded at sea.

Another consideration is that *P. argus* is abundant in the Western Central Atlantic from northeastern South America, throughout the Caribbean, and as far north as Bermuda (Silberman et al. 1994). As a result, altered abundance may have widespread effects. The distribution of a closely related sub-species or species (*Panulirus argus westoni*) extends further south into southern Brazil (Sarver et al. 1998, Tourinho et al. 2012). Like all spiny lobsters (Palinuridae), *P. argus* has a long planktonic larval duration, in this case lasting 8 months (Goldstein et al. 2008) followed by the settlement of postlarvae in coastal vegetation (including seagrass and macroalgae), and metamorphosis into an asocial early benthic stage that dwells in nursery vegetation. Juvenile lobsters continue to dwell in shallow nursery areas then migrate to coral reefs in deeper waters, where the long-lived adults dwell. Individuals become social a few months after settlement and this gregariousness persists for the rest of its juvenile and adult life (Butler et al. 1997, Briones-Fourzán et al. 2006). Gregarious behavior is common among Palinurids and has been observed in both juvenile and adult life stages of many species (Berrill 1975). Chemical cues attract conspecifics and thus enable this social behavior (Ratchford and Eggelston 1998, Butler et al. 1999, Nevitt et al. 2000, Horner et al. 2006a). The reasons for the evolution of social behavior in spiny lobsters have been debated with evidence supporting its advantage for group defense (Butler et al. 1999) and for locating shelters housing conspecifics...
more quickly (i.e., the “guide effect”); (Childress and Herrnkind 1997, 2001). This tendency to aggregate has led to the large numbers of Caribbean spiny lobsters that are observed at casitas.

The long planktonic duration of *P. argus* larvae also has relevance to managers in the wider Caribbean who should take into consideration evidence that the species is a single, panmictic stock in the Western Atlantic. If so, it therefore should be managed as a single unit, not separate stocks defined by geopolitical boundaries. Evidence of this is seen in the mitochondrial DNA linkages that span from Bermuda to Brazil (Silberman et al. 1994, Naro-Maciel et al. 2011), as well as the results of biophysical modeling of *P. argus* larval dispersal in the Caribbean (Kough et al. 2013). Yet, there are subregions within the Caribbean where strong, persistent oceanographic features tend to retain *P. argus* larvae near their natal source and thus enhance local retention of larvae (Butler et al. 2011, Kough et al. 2013). This appears to be the case for The Bahamas (Kough et al. 2013). Thus, there is an opportunity for management within The Bahamas to impact the health of the fishery directly within its waters with little dependence on spawning biomass and recruitment from other territories. This independent response to management and fishing pressure, by some definitions, justifies that The Bahamas should be considered a stock (Carvalho and Hauser 1994).

Of relevance to the issue as to whether casitas alter the sustainability of lobster fisheries is the broader debate about whether the observed biomass of organisms at artificial structures in the marine environment is due to attraction verses production (Alevizon and Gorham 1989, Polovina and Sakai 1989, Pickering and Whitmarsh 1997). The number or weight of organisms associated with artificial structures might simply be due to the concentration of organisms resulting from their attraction to artificial structures. Alternatively, the biomass observed at artificial structures might be due to an increase in biomass of the particular species in the overall community, especially if adequate refuge was a major limiting factor. The same disputation applies to lobsters found at casitas (Briones-Fourzán et al. 2007, Gutzler et al. 2015). If casitas result in increased production then there is a chance that they enhance the lobster fishery. However, a broader suite of environmental impacts such as physical damage to the environment would have to be considered to confirm this conclusion. If casitas simply attract (and aggregate) lobsters then the increase in efficiency for fishers makes overfishing more possible. It should also be borne in mind that excessive fishing pressure may negate the effects of increased production (Pickering and Whitmarsh 1997). The uncertainty concerning attraction or
production, and the possibility that other negative consequences of casita use may exist, are widely recognized among Caribbean lobster fishery managers (FAO 2006), some of whom urge caution in adopting this gear. Evidence of this is seen in the hortatory St. George’s Declaration on Conservation, Management and Sustainable Use of the Caribbean Spiny Lobster (Panulirus argus) where, among other tenets, Caribbean countries have agreed that research should be undertaken to determine the effects of casitas. Though nonbinding, this applies to countries like The Bahamas that have already introduced casitas and to countries that seek to introduce them.

Casitas and Lobster Disease

The social nature of *P. argus* leads to the aggregation of lobsters in casitas in unnaturally high numbers. Although diseases are a natural occurrence and a major component of the natural mortality for juveniles (Cruz et al. 2007), it is well documented that overcrowding increases the prevalence of disease (Elton 1931, Murray and Peeler 2005). Overcrowding may also lead to resource constraints such as limited food availability. Higher disease rates and lowered food availability per lobster can individually and simultaneously lead to slower growth rates and higher mortality rates. Higher mortality rates might also be further induced if predators are attracted to casitas. Mortality and growth rates are key factors that directly affect fishery productivity. Fisheries managers and stock assessment modelers commonly seek to estimate (Leocadio and Cruz 2008) and augment growth and mortality rates in an effort to achieve quantitative sustainability targets.

There are two common diseases that occur in *P. argus* populations and the transmission of both is correlated with contact of infected conspecifics: shell disease is caused by bacteria and PaV1 by a viral pathogen. Shell disease occurs in a number of species of crustaceans (Porter 2004, Davies et al. 2014). The occurrence of shell disease in *P. argus* was not documented until 1998, but it has been observed in *Homarus americanus* (H. Milne-Edwards 1847) for nearly a century (Porter 2004). In *P. argus* and *H. americanus*, shell disease has been attributed to seven groups of bacteria with six common to shell disease lesions of both species; one of the seven types of bacteria are found only in *P. argus* (Porter 2004). The groups flavobacteriaceae and *Vibrio spp.* (Vogan et al. 2002) have been specifically implicated in the disease (Davies et al. 2014) and physical damage to the cuticle of the lobster is considered a major inducer of shell disease (Delves-Broughton et al. 1976, Porter 2004). However, the factors that influence
susceptibility to shell disease are not fully understood and environmental triggers may play a role in lobster vulnerability to the disease (Delves-Broughton et al. 1976, Vogan et al. 2002). Although only severe infections are lethal and the prevalence of the disease in *P. argus* populations in Florida is very low (Porter et al. 2001), a shell disease epizootic infecting *H. americanus* in New England has resulted in major economic losses (Castro and Angell 2000, Davies et al. 2014).

PaV1 is more of an immediate threat to *P. argus* survival and fishery sustainability in the Caribbean. Soon after this lethal viral disease was first documented in Florida in 2000 (Shields and Behringer 2004) it was discovered that smaller lobster were more susceptible to infection with prevalence decreasing as size increases (Behringer et al. 2011). The virus is transmitted via close contact with infected individuals or over short distances in seawater (Butler et. al 2008), thus the crowding of lobsters within casitas has been a cause of some concern (Behringer and Butler 2006, Briones et al. 2012, Gutzler et al. 2015). However, *P. argus* appear to have evolved a natural quarantine-like behavior that isolates infected individuals (Behringer et al. 2006) and appears to reduce the probability of epizootics (Butler et al. 2015). Although perhaps counterintuitive, it is also possible that the addition of appropriately sized and placed casitas could help to alleviate habitat limitation, thereby reducing crowding and the risk of PaV1 transmission in natural shelters. Regardless, the risk of overfishing lobster stocks is exacerbated if the effects of the PaV1 disease and its potential interaction with gear type go undetected or unaccounted for by stock assessment modelers.

*Casitas and Predation*

Casitas may potentially alter patterns of lobster mortality if they contribute to a higher incidence of injury, disease, or size-specific predation. Injuries can be consequential to lobsters if they result in a higher risk of disease, slower growth, or greater probability of predation. Overcrowding in casitas may result in higher rates of injury. Infection by PaV1 is uncorrelated with injuries, but shell disease is more likely in lobsters with abraded cuticles. The effects of injuries on lobster growth are inconsistent. For example, the growth of *Panulirus gracilis* (Streets 1871) and *Panulirus inflatus* (Bouvier 1895) in Mexico (Briones-Fourzán and Lozano-Álvarez 2003) is unaffected by injury, whereas injury reduced growth rates in male *Jasus edwardsii* (Hutton 1875) in Tasmania (Emery et al. 2016) and *P. argus* in Florida (Hunt and
Altered growth rates have implications for fishery sustainability because the reliability of fishery yield estimates is heavily dependent on knowledge of growth rates (Punt et al. 1997). Understanding the effect of injuries on crustaceans and the subsequent effects on growth across a population is thus critically important if sustainable harvest targets are to be considered reliable (Emery et al. 2016).

**Casitas and Prey Availability**

Reduced growth rates in *P. argus* may also result if food is limited or locally depleted, as occurs near casitas (Nizinski 2007). This concern applies especially to smaller lobsters that spend most of their time near and within their refuge (Weiss et al. 2008). Juveniles generally exhibit high residence behavior (Cruz et al. 2007) and are known to leave casitas 2-30 times per night for a maximum of 10 minutes per occasion, depending on their state of hunger and risk of predation (Weiss et al. 2008). If enhanced risk-taking spurred by food limitations is a common occurrence near casitas, population wide higher mortality rates for juveniles might also befall populations targeted by casitas with consequent implications for safe harvest levels and sustainability.

All things considered, if casitas affect disease rates, mortality rates and growth rates, there are implications for the accuracy of stock assessments and fishery sustainability. To complicate matters further, there may also be interactions between these effects. These have a small potential to be positive as with the earlier example of predation reducing the presence of diseased animals thereby reducing chances of disease transmission. There is also a possibility that there are synergistic but negative interactions. Managers must also bear in mind that an assessment of sustainability also goes beyond biological effects on target species. Among the additional biological considerations is size selectivity of fishing gear, bycatch and the ability to conduct accurate stock assessments.

**Casita and Trap Bycatch and Size Structure Considerations**

Consideration of bycatch and selectivity are major issues that fisheries managers must address (Hall et al. 2000, Davis 2002, Harrington et al. 2005) and reducing bycatch mortality is considered a key aspect of fisheries management (Uhlmann and Broadhurst 2015). Even though there is awareness of the need to address bycatch, there is often great uncertainty about the
amount of bycatch associated with specific fishing practices (Lewison et al. 2004, Zeller and Pauly 2005). This is compounded by the perception that some components of bycatch are of no economic value, which does not negate the possibility that such bycatch may have far reaching negative ecological repercussions (Crowder and Murawski 1998, Harrington et al. 2005). For example, of major concern is the fate of the "live discards" component of bycatch (Davis 2002). Live discards may not survive, which impacts stock assessment accuracy and, by extension, the likelihood of sustainability (Uhlmann and Broadhurst 2015). A related ecological concern is the unintended damage to the environment caused by lost gear or movement of gear (Lewis et al. 2009, Arthur et al. 2014). "Ghost fishing" by lost fishing gear such as traps continue to attract target and non-target species, many of which then perish (Bilkovic et al. 2012). These effects on targeted and untargeted taxa often go unrecorded and unquantified. Disparities in the amount and character of bycatch potentially exist between the two primary types of fishing gear (i.e., casitas and wooden-lathe traps) that are utilized in the Bahamian lobster fishery and until now this issue has not been investigated.

Despite the shortcomings that may result from using fishery dependent data, these are often the only data available to managers (Punt et al. 2013). This is especially so when size frequency data are utilized to estimate key parameters such as fishing mortality and biomass (Punt et al. 2014). Fishery-dependent size frequency based or size frequency supplemented lobster stock assessments have been attempted or conducted for The Bahamas and will continue for the foreseeable future (CRFM 2011). Although it is recognized that lobster size-frequencies determined from fishery catch in The Bahamas may be biased, the extent of the biases is unknown. This dissertation also examines the size frequencies of lobsters in relation to the use of casitas and traps.

Monitoring Casita Distribution and Abundance

Another issue that fisheries managers wrangle with in The Bahamas is the ability to monitor the deployment of gear by the lobster fishery on the expansive (>116,000 km²) and often remote, shallow banks of The Bahamas. This presents challenges for the collection of adequate data with which to conduct stock assessments on a scale representative of the Bahamas. To do so, one must determine the number and spatial distribution of casitas. One of the options for detecting the location, enumerating the number, and revealing the spatial distribution of casitas is
through the use of remote sensing. Remote sensing allows the repeated collection of data about objects, areas, and locations from afar (Green et al. 1996, Lillesand et al. 2014). Thus, it offers an advantage for long-term monitoring of remote or hard to reach areas (Campbell 1996). Although remote sensing can be expensive, the cost of traditional survey methods can also be prohibitive (Mumby et al. 1999), especially over large or remote areas such as The Bahamas. This enhances the appeal of remote sensing. As the spatial resolution available through modern day remote sensing allows the detection of objects as small as one meter in size (Inglada 2007, Cheng and Han 2016), its appeal for detecting typical 1.8m x 1.2m casitas used in The Bahamas is evident. Previous studies in The Bahamas have also demonstrated that it is possible to monitor Bahamian bathymetry. This includes important foraging and settlement areas for lobsters such as seagrass (Louchard et al. 2002, Dierssen et al. 2003).

Dissertation Objectives

In Chapter II, I examine the possibility that casitas will affect the rates of growth, mortality, and injury as well as the incidence of disease in *P. argus* in the Bahamian lobster fishery. I hypothesize that small lobsters (e.g., those < 80 mm carapace length) dwelling in casitas will be exposed to higher mortality from predators that are also attracted to casitas, and they will be in poorer condition, will grow slower, and will be more prone to disease due to overcrowding. I explore the effects of bycatch and the implications of the size selectivity of lobsters associated with casitas and lobster traps in Chapter III. I hypothesize that juvenile lobsters collected from casitas by fishers using hooks and then released will experience higher mortality and different size-structures than those caught and released from traps. In contrast, I project that traps have a higher bycatch of other taxa compared to casitas, but minimal effect on undersized lobsters. Finally, in Chapter IV I examine the feasibility of using remote sensing to ascertain the number and distribution of casitas. My hypotheses in this case are that casitas can be detected in various benthic habitats common in The Bahamas, and therefore the distribution and abundance of casitas can be estimated using remote sensing technology after proper calibration and ground-truthing.
CHAPTER II
THE EFFECT OF CASITAS ON PANULIRUS ARGUS MORTALITY, GROWTH, AND SUSCEPTIBILITY TO DISEASE IN THE BAHAMAS

INTRODUCTION

Fishery productivity and sustainability have a variety of biological influences, many of which fisheries managers attempt to measure, monitor, or control. Key life history parameters such as mortality, growth, recruitment, and reproduction (Jones and Wells 1998, Leocadio and Cruz 2008, Kilduff et al. 2009) are often among the biological influences that managers attempt to measure, control, or utilize in stock assessments (Phillips and Melville-Smith 2006, Miller 2009). Mortality, whether it is natural mortality or due to exploitation (fishing mortality), results in the loss of biomass from a population. This is in contrast to growth and recruitment, means by which biomass increases (Kilduff et al. 2009). The balance of these additions and removals of biomass ultimately determines whether population sizes increase or decrease and the rate at which these changes occur. This balance includes instances when higher mortality results in faster growth rates as may occur when the exploitation of older age classes is allowed in order to reduce the proportion of slower growing or less fecund individuals in favor of faster growing younger individuals (Miller 2009).

Growth and mortality rates strongly influence the sustainability of fisheries and these influences can be complex. Errors in the estimation of growth and mortality, or the absence of estimates of these parameters, have potentially far reaching consequences for the accuracy of stock assessments and resulting management advice (Maunder and Piner 2015). Knowledge of growth and mortality rates has also traditionally played an important role in the identification of stocks or management units (Pawson and Jennings 1996, Begg et al. 1999b, Begg and Waldman 1999). Growth rates also influence how quickly minimum size limits are attained (Kilduff et al. 2009). Repercussions concerning the accuracy of growth and mortality estimates make awareness of factors that influence them particularly important. Some of these factors include the effects of diseases and food availability as these may result in slower growth and higher mortality (Lightner and Redman 1985, Smith et al. 2003). Further, knowledge of influences on
growth and mortality can be a key step in the mitigation of negative impacts. For example, awareness that the use of traps may result in ghost fishing has led to the use of biodegradable materials in traps to reduce the length of time traps will fish should they get lost (Valdemarsen and Suuronen 2003). Similarly, circle fish hooks are used to minimize injury and mortality rates of non-target pelagic fish compared to J-hooks (Kerstetter and Graves 2006) and turtle excluder devices help to reduce the mortality of turtles resulting from the use of trawl nets (Valdemarsen and Suuronen 2003). Many more examples of measures implemented to meliorate unintended fishing mortality or unintended impacts on growth rates exist. By comparison, little is known about the effects of casitas on growth, mortality and on other aspects of the sustainability of lobster fisheries. Casitas are open, table-like structures under which Caribbean spiny lobster (P. argus Latrielle 1804) congregate and can be rapidly captured by divers (see Figure 2).

Investigations concerning effects on health and mortality due to casitas are particularly important because in recent decades several fisheries in the Caribbean have transitioned from the use of traps to the use of casitas. This includes The Bahamas where casitas became the main fishing gear used in the lobster fishery since the mid-1990s.

The use of casitas to target P. argus has become popular because large numbers of P. argus aggregate under them for refuge. These aggregations are a natural occurrence that results from strong sociality among conspecifics (Lewis et al. 1952, Kanciruk 1980, Marx and Herrnkind 1985, Herrnkind and Butler 1986, Butler et al. 1997) that is believed to be for group defense (Butler et al. 1999) and for locating shelters housing conspecifics more quickly (i.e., the “guide effect”); (Childress and Herrnkind 1997, 2001). The increase in catchability that results from this tendency to aggregate is well known among fishers that utilize casitas and those that continue to use traps (FAO/WECAFC 2007). In some countries, including Cuba and Mexico, casitas have been utilized for decades and there is a strong and growing interest in their use throughout the wider Caribbean (Ramos-Aguilar 2003, FAO/WECAFC 2007). However, they are currently banned in other countries, such as the United States. The widespread change in fishing gear from traps to casitas has raised questions about the potential effects on the environment and the sustainability of P. argus stocks, and has also stimulated debate as to whether the observed biomass of lobsters in casitas (i.e., artificial structures) is due to attraction verses production (Alevizon and Gorham 1989, Polovina and Sakai 1989, Pickering and Whitmarsh 1997, Gutzler et al. 2015). If casitas result in increased lobster growth or survival,
then they may enhance lobster stocks. However, if they simply aggregate lobsters that are normally more widely scattered in the environment, then they could be deleterious to lobster stocks if concentrations of lobster become food-limited, attract predators, increase disease rates, increase the efficiency of capture by the fishery or if slower growth results. The uncertainty concerning attraction or production, as well as the possibility that there may be other negative consequences of casitas are widely recognized among Caribbean lobster fishery managers (FAO/WECAFC 2007) and have spurred a sense of caution in adopting their expanded use.

A number of potentially deleterious effects on lobsters may occur because naturally occurring aggregations of lobsters are rarely as large as those under casitas. For example, diseases are a natural occurrence and a major component of the natural mortality for juveniles (Cruz et al. 2007), but overcrowding increases the transmission of disease in social animals (Elton 1931, Murray and Peeler 2005). Overcrowding may also lead to resource constraints such as limited food availability. In turn, a higher prevalence of disease and lower food availability may result in slower growth and higher mortality (Lightner and Redman 1985, Smith et al. 2003). It is possible that predation risk is lower at casitas where natural shelter for lobsters is limiting (Sosa-Cordero et al. 1998), but it is also possible that predators may be drawn to casitas because of their structure or the scent of large aggregations of lobsters and thus impart higher rates of mortality. Indeed, olfactory cues play a major role in the long-range detection of prey by sharks (Hodgson and Mathewson 1971, Gardiner et al. 2014) including nurse sharks (*Ginglymostoma cirratum* (Bonnaterre 1788)), a major lobster predator. Concentrations of large lobsters under casitas are also likely to attract smaller and more vulnerable lobsters away from more appropriate sized dens, resulting in higher predation on juvenile lobsters (Schratweiser 1999, Gutzler et al. 2015)

Among the diseases of greatest concern for *P. argus* fisheries and whose transmission is at least in part density-dependent are shell disease (a bacterial infection) and PaV1, the only known viral disease in lobsters and which typically results in death for juvenile lobsters. Shell disease occurs in a number of species of crustaceans (Porter 2004, Davies et al. 2014). It was observed nearly a century ago in the American clawed lobster (*Homarus americanus*) (Porter 2004), but the emergence of a shell disease epizootic in *H. americanus* in New England, first documented in 1998, has resulted in major economic losses along the east coast of the USA where it threatens fisheries (Castro and Angell 2000, Davies et al. 2014). The occurrence of shell
disease in *P. argus* was not documented until 1998 (Shields 2011) and its natural prevalence appears low, except under crowded conditions (Porter 2004). Though physical damage to the lobster cuticle is considered a major facilitator of shell disease (Delves-Broughton et al. 1976, Porter 2004), the triggers for susceptibility to shell disease are not fully understood. As such, shell disease can be considered a potential threat to *P. argus* fisheries because of its potential impacts on lobster mortality, fecundity and marketability (Cobb et al. 2006) and its prevalence could be effected by injuries or patterns of aggregation associated with the type of gear used in a fishery.

PaV1 may be more of an immediate threat to the sustainability of *P. argus* fisheries in the Caribbean. The lethal, pathogenic virus was first documented in Florida in 2000 (Shields and Behringer 2004, Behringer et al. 2011) but has since been found in lobsters throughout much of the Caribbean (Moss et al. 2013). Transmission of the virus occurs through several routes (e.g., waterborne, cohabitation, consumption) and juvenile lobsters are more susceptible to PaV1 infection, suffering > 90% mortality once infected (Butler et al. 2008). The mortality rates observed in juveniles have major implications for the sustainability of lobster fisheries. As these juvenile lobsters are typically off limits due to management imposed minimum size limits, they have a strong negative effect on the fishery when afflicted with PaV1 as they die before they recruit to the fishery and they do so before they make a reproductive contribution to the population (Behringer et al. 2012).

Fortunately, there is evidence that avoidance behavior exhibited by healthy conspecifics may curtail the spread of the virus in nature (Butler et al. 2008). Nonetheless, as waterborne transmission of infections can occur over a few meters and transmission can occur through close contact among lobsters (Butler et al. 2008), the aggregations of lobsters found under typical sized casitas (1.8m x 1.2m) used in the Bahamian lobster fishery might enhance natural rates of disease transmission. This danger of disease transmission in casitas could affect lobsters as small as 15-20mm carapace length (CL) as gregarious behavior and attraction to conspecifics ensues at that size (Butler et al. 2006).

Overcrowding of lobsters under casitas also has the potential to result in higher rates of injury and subsequently slower growth, as observed when male *J. edwardsii* are injured in Tasmania (Emery et al. 2016). Altered growth also has implications for fishery sustainability because the reliability of estimates of fishery yield depend on knowledge of growth rates (Punt et
Detecting injuries that result from lobster association with casitas is an important step in discerning the effects of casitas on growth in comparison to alternative fishing methods such as traps (Hunt et al. 1986).

Reduced growth of *P. argus* may also result if food is limited or locally depleted. This concern applies especially to smaller lobsters. Whereas adults may venture greater distances to feed at night (Cox et al. 1997), juveniles spend most of their time within the immediate vicinity of their refuge (Weiss et al. 2008) making them more susceptible to food depletion near casitas (Nizinski 2007). Juveniles generally remain close to shelter (Cruz et al. 2007) and leave casitas 2-30 times per night for a maximum of 10 minutes per occasion, depending on their state of hunger and risk of predation (Weiss et al. 2008). This increased risk taking spurred by food limitation potentially increases mortality on juvenile lobsters.

There may also be higher mortality associated with lobster use of casitas resulting from the attraction of predators to those same structures. This risk is highest for the smallest of lobsters as they are normally subjected to higher predation than larger animals (Smith and Herrkind 1992). Higher predation has indeed been observed in *P. argus* sheltering under casitas under certain circumstances. Gutzler et al (2015) found that juvenile lobsters experienced higher predation when associated with casitas deployed in nursery habitats in Florida, but observed no change in mortality for larger lobsters. If this pattern is widespread, it could limit recruitment across the population. All things considered, if the use of casitas by fishers has resulted in higher lobster mortality either due to diseases, predation, injuries, or slower growth, then there are consequences for the accurate estimation of safe exploitation levels.

These issues are all of concern for The Bahamas where most lobsters are caught using casitas. The lobster fishery is the primary commercial fishery in The Bahamas and is responsible for approximately 88% of recorded landings by value and 63% of recorded landings by weight (Gittens 2008). Wooden lathe traps are also used in the fishery, but based on records of the Bahamas Department of Marine Resources, licenses were issued for approximately 48,000 lobster traps during the 2015-2016 season, whereas private researchers have estimated that the number of casitas in the Bahamas now ranges from hundreds of thousands to over 1 million (Doerr 2014).

Similarly, casitas have gained increasing popularity with fishermen (FAO/WECAFC 2007) in many parts of the Caribbean where landings of *P. argus* are valued at over $800 million
per annum (FAO/WECAFC 2007) with a production of 35,642 metric tons (FAO 2015). In The Bahamas, 6088 metric tons (converted to whole weight) of lobsters were landed in 2013 valued at $47.7 million, making it the single largest and most valuable fishery for *P. argus* in the Caribbean. Across the wider Caribbean, the lobster fishery generates foreign exchange, provides direct employment to 33,000 fishers (Chavez 2009), indirect employment to local people, and has additional but unknown value for subsistence fishing, recreational fishing, and multiplier effects such as tourism (FAO 2007). Sustainable use of this fishery thus has repercussions for Caribbean economies, food security and quality of life.

In this chapter, I examine ecological issues that must be considered if sustainable management of a casita-based lobster fishery is to be realized in The Bahamas. Specifically, I hypothesize that small lobster (e.g., those < 80 mm CL) dwelling in casitas will be:

(a) exposed to higher mortality from predators that are also attracted to the structure afforded by casitas, and

(b) in poorer condition and thus experience lower growth and more disease due to overcrowding.

**METHODS**

The fishing grounds selected as sampling sites included commercial fishing areas in Sandy Point and Marsh Harbour, Abaco (Figure 1). These areas are representative of typical back reef lagoon (i.e., Marsh Harbor site) and shallow bank (i.e., Sandy Point) areas where casitas and traps are utilized in The Bahamas. Sampling trips took place during the closed and open seasons. It was important to include samples from the closed season (April 1 – July 31) because commercial fishermen in the Bahamas typically leave casitas in the water year-round, as is also the case for an unknown proportion of the commercial traps. Density-dependent health effects may be exacerbated for lobsters residing in casitas or traps during the four month closed season more so than during the open season when gear is regularly fished and lobster densities within them are lower.

At both study sites, I deployed six experimental casitas and 10 experimental wooden lathe traps built to the specifications of typical commercial lobster fishing gear utilized in The
Bahamas. Casitas in the fishery vary in size and materials used for construction, but most are 1.8m long x 1.2m wide with a top made of corrugated metal and sides composed of 1.8m lengths of 5cm x 15cm lumber that raises the top approximately 15cm off the seafloor (Figure 2). Both ends of casitas remain open to permit ingress and egress of lobsters. Data from my experimental casitas were sometimes supplemented by sampling from fisher casitas of the same approximate dimensions, although the sizes and construction materials of fisher casitas vary (e.g., car hoods for tops and sections of pine trees for the legs).

Consistent with commercial lobster trap requirements outlined in The Bahamas’ *Fisheries Resources (Jurisdiction and Conservation) Regulations* (1986), the experimental traps used in this study were 0.9m long x 0.6m wide x 0.6m tall with wood slats no less than 2.54 cm apart; each was weighted with a concrete slab on the bottom.

Experimental casitas and traps were placed in or near seagrass in water 3-5m deep, a common practice of fishers. At both study sites, casitas were placed approximately 90m apart in a single line. Traps were also placed in a line approximately 90m apart from one another and within 90-300m of the line of casitas. The distance between traps and casitas was similar to those deployed by fishers in the study areas. Like the commercial fishery, traps were left at sampling sites throughout the season but unlike commercial traps whose lids are nailed shut, I fixed the trap lid open when I was not sampling them. This was done to prevent lobsters from being trapped between sampling periods. I baited and closed the trap lids 10-14 days before each sampling period. This baiting and sampling interval was to ensure that the traps were conditioned to attract lobsters and to mimic, as far as reasonable, the length of time fishers leave traps in the water during the open season. Fishers report that traps must remain in the water to become “seasoned” for a few weeks before they attract lobsters.
Figure 1: Map of The Bahamas showing the location of sampling sites in Marsh Harbour and Sandy Point near Abaco island.
Figure 2: Photo of an experimental casita on the seafloor near Abaco, The Bahamas. The casita is 1.8m x 1.2m with 1.8m long wood used as legs along the longer sides of the casita. The 1.2m sides were left open allowing lobsters and other organisms to enter and leave. Two cinder blocks were used as weights. Sand and algae covered the casita a few months after deployment. Surrounding the casita is sand and sparse patchy seagrass.

I also attempted to sample lobsters from natural shelters within each study area to compare their characteristics with those collected from experimental gear. However, few lobsters were found in the natural environment in the vicinity of sampling gear despite repeated dives by teams of divers. I chose not to collect lobsters from natural areas beyond the vicinity of sampling gear to avoid potential differences in lobster health and growth attributable to habitat. Due to the very low availability of lobsters in natural shelters, comparisons of lobster health characteristics were limited to those collected in casitas and traps.

**Health and Injury Comparison**

Lobsters were collected from traps and casitas from both sampling areas between July 2012 and February 2016 to compare a variety of measures of lobster health and growth. For each lobster, visible signs of PaV1 (as described by Shields and Behringer 2004), sex, carapace length
and number of injuries were recorded. In addition, the blood refractive index (BRI), a measure of the blood protein index (BPI) of lobster haemolymph, was recorded for each lobster sampled with the aid of a refractometer (Reichert Industrial Fluid Tester 40 model 137540 LO). BPI/BRI is considered to be representative of lobster physiology and health condition in crustaceans (Moore et al. 2000) and lobsters (Chandrapavan et al. 2011), including *P. argus* (Behringer and Butler 2006a).

In addition, a pleopod sample was collected from each lobster for microscopic determination of molt stage (Lyle and MacDonald 1983). Differences in growth rates in lobsters and other crustaceans are influenced by the size of molt increments and intermolt interval (Forcucci et al. 1994, Ehrhardt 2008). As such, lobsters growing at a faster rate are less likely to be in an intermolt stage. This allowed molt stage readings to be used to discern differences in molt frequency between lobsters associated with each gear type, thus provided a relative measure of differences in lobster growth rates collected from each gear type (Gutzler et al. 2015, Gutzler and Butler 2017).

Lobsters are known to travel 25-400m overnight when they move between casitas in Mexico (Ramos-Aguilar et al. 2003). To avoid sampling lobsters more than once due to movement between fishing gears, lobsters that were sampled in neighboring casitas or traps on consecutive days were observed for the presence of a clipped pleopod as an indication that they were already sampled.

A logistic regression was conducted to ascertain whether molt stage was significantly affected by fishing gear or by other variables such as: carapace length (as a continuous variable), season (open vs. closed), the interaction between fishing gear and carapace length, and the interaction between fishing gear and season as independent variables. These independent variables were also utilized in a logistic regression that instead utilized injuries (presence or absence) as the dependent variable to determine whether there were any significant effects on injuries. A log linear analysis was also utilized due to shortcomings of the logistic regression (as detailed in the Results section). Because molt stage also influences blood constituent composition (Mercaldo-Allen 1991), it too was considered in our analyses of blood protein in *P. argus* (Travis 1955). A log linear analysis also permitted the detection of interactions between gear type and BPI, as well as the effect of gear type on the interaction between molt stage and BPI.
As a log linear analysis requires that all variables be categorical, carapace lengths were categorized into two groups (lobsters < 80mm CL and those ≥ 80mm) based on maturity estimates for this species. Size at maturity varies by location in spiny lobsters including *P. argus* (Sutcliffe 1952, Kanciruk and Herrnkind 1976, Ehrhardt 2005). One of the smallest reproductive females recorded in The Bahamas was 74mm CL (Kanciruk and Herrnkind 1976) and 50% of lobsters are estimated to be mature around 85mm CL (Ehrhardt 2005), whereas in neighboring waters (Florida, USA) (where 76mm CL is the minimum size limit) most reproductive *P. argus* are in the size class 75-85 mm CL (Gregory Jr et al. 1982). Bearing in mind uncertainties in the way size at maturity has been estimated in Palinurids and that fishing pressure may have an influence (Kanciruk 1980), the size of mature lobsters in The Bahamas is likely 74-85 mm CL.

In the case of BPI, the range of BPI values (3-23) was treated as low (3-10) and high (11-23). Molt stages were divided into intermolt and molting categories and injuries into non-injured and injured. Fishing seasons were based on Bahamian law and included the closed season (April 1-July 31) and open-season (August 1-March 31). Potential seasonal effects were investigated to determine whether any negative effects associated with casitas might only emerge during the closed season when lobsters might spend more time in casitas and at higher densities. This resulted in a contingency table with 64 cells comprised of CL (2 categories) x BPI (2) x molt stage (2) x injuries (2) x location (2) x season (2). Thus, a minimum of 384 observations was required to meet the minimum standard of a mean of 6 expected observations per cell.

*Starvation Experiment*

Wooden lathe traps were the main fishing gear utilized in The Bahamas until the 1990s when the use of casitas became commonplace, though unregulated (Gittens 2008). Now, it is believed that the vast majority of fishers use casitas even though traps remain as a viable, legal alternative. If casitas were deemed deleterious to the sustainability of the fishery and were banned, fishers would likely return to using traps. So an evaluation of the effects of casitas on lobster populations in comparison to traps is necessary.

Ghost fishing due to trap loss is a well-documented drawback to trap use (Butler and Matthews 2015), another more controllable impact is the effect of simply not removing traps from the water during the closed season. While fishers remove traps during the closed season, it is not known how many traps are left at sea in The Bahamas. These traps are not lost and fishers
are not actively fishing them, but lobsters enter them nonetheless and remain trapped for various lengths of time. During the fishing season, fishers typically empty lobsters from traps every 3-21 days, depending on the season and weather. However, traps left in the water during the closed season may constrain lobsters for up to 16 weeks, the length of the closed season. My starvation experiment involved the placement of lobsters (n = 93) in traps for 6 weeks or more. Data collected included: carapace length, BPI, injuries, and sex as well as observations on shell condition, and shell disease. Pictures of lobsters were taken in some instances to monitor shell disease progression. There were two iterations of this experiment. During the first, observations were collected in Sandy Point and Marsh Harbour. A total of 72 lobsters were placed in 10 traps at each site during January 2015 and July 2015. Traps were not sealed but lobsters were tagged to discern experimental lobsters from new entrants. Lobsters were then sampled approximately every three weeks for six weeks in all instances and up to twelve weeks in some instances. A Chi-square analysis was conducted to determine whether there was a significant relationship between the presence of shell disease and starvation time.

Due to low sample sizes at six weeks and 12 weeks in addition to logistical challenges with maintaining the necessary sampling frequency, a second iteration of this experiment was conducted in New Providence, The Bahamas. The lobsters utilized in New Providence were obtained from Cay Lobos from a commercial trap fisherman during November 2015. During this experiment 21 lobsters were randomly interspersed between three sealed traps. Data were collected at the beginning of the experiment and at three weeks and six weeks. A 1-way repeated measures ANOVA with follow-up pairwise comparisons was conducted to ascertain whether there was a change in disease prevalence within subjects across time.

Predation Comparison

I evaluated the risk of predation for lobsters tethered for 24 hrs at casitas versus nearby natural shelters at both sample sites (Marsh Harbor, Sandy Point) during June 2014-February 2016. A total of 120 lobsters (size range: 21-111.5 mm CL) collected from casitas in the area were tethered, but after accounting for escapes (i.e., no obvious signs of predation), 109 lobsters were included in the tethering analyses. Tethers were comprised of 2m of monofilament line tied and glued to the dorsal side of the carapace of each lobster either between the 2nd and 3rd or 3rd and 4th pairs of legs. The other end of the monofilament was tied to a weight (e.g., stones, bricks
or empty conch shells) heavy enough to keep the lobster in place. Weights were placed near enough to casitas or natural shelters to allow the tethered lobsters to obtain refuge underneath the structure. Casitas were chosen that had additional lobsters present and one lobster was tethered at each entrance to the casita; thus, two lobsters per casita. Within 10-30m of each casita, two lobsters were also tethered individually in natural shelters (e.g., near holes, rocks, coral heads, sponges, undercut edges of seagrass, dense seagrass patch). After 24hrs, the presence or absence of each lobster was noted and survivors released. These data were analyzed using a logistic regression with carapace length treated as a continuous variable.

Many have commented on the advantages and limitations of the tethering approach to measuring predation, noting its usefulness as a relative measure of predation but not necessarily as a measure of true predation rates (Smith and Herrkind 1992, Mintz et al. 1994). The possibility of biases exist because tethering may also increase the success of predators, depending on the predators involved (Mills et al. 2008). It is thus advisable to incorporate consideration of predator assemblages when evaluating tethering results (Mills et al. 2008). Therefore, to augment my assessment of predation risk for lobsters in casitas and natural shelters, my tethering data were supplemented by still photos.

Predation risk at casitas verses nearby natural shelters was also evaluated by collecting still pictures with GoPro cameras deployed at Marsh Harbor and Sandy Point between May 2014 and February 2016. Cameras were placed near casitas with lobsters and near natural shelters; tethered lobsters were used to simulate lobster presence in the natural shelters. A single camera was placed within 3m of each casita or natural den. I made 51 separate camera deployments, each lasting 2–4 hrs with the camera set to collect 10mp pictures in 30 sec intervals. Cameras were set in the afternoon and collected on the same evening or the next day.

The presence, number, and species of lobster predators and apex predators in each frame were noted among the 17,659 pictures that were collected. Only predators previously documented in peer-reviewed literature were categorized as lobster predators. Where the same predator was observed in sequential frames, the predator’s presence was recorded as a separate predation risk occurrence as the length of time a predator is present also reflects predation risk. In addition, each predator in frames containing multiple predators was individually scored as this also reflects a higher predation risk. A mixed model 2-way ANOVA was used to analyze these
data with treatment (casita and natural environment) as the fixed factor and site as the random factor.

RESULTS

Health and Injury Comparison

A total of 634 lobsters were collected during this study but none presented with visual signs of PaV1, thus no late stage infections were detected. This included juvenile and adult lobsters.

A logistic regression showed that there were no significant effects on molt stage resulting from fishing gear (p = 0.195, df = 1), carapace length (continuous variable; p = 0.491, df = 1), season (p = 0.340, df = 1), the interaction between fishing gear and season (p = 0.569, df = 1) nor the interaction between fishing gear and carapace length (p = 0.117, df = 1). However, an analysis of the effects of these factors on the presence of injuries was inconclusive as an Omnibus test of model coefficients showed that a logistic regression model did not fit the data adequately ($\chi^2 = 5.959, p = 0.310, df = 5$). As a result, in lieu of a logistic regression, a log-linear analysis was conducted but required that carapace length be treated as a categorical variable.

Also of consideration was the need to test for the effects of the aforementioned factors on BPI. Though a multiple linear regression would be more ideal to detect the effects on BPI because it was a continuous dependent variable, BPI was also included in the log-linear analysis as a categorical variable. This was to reduce the chance of a Type I error because much of the data included in the analysis were already analyzed as a part of the logistic regression utilized to test effects on molt stage.

The log linear analysis revealed that there were significant 3-way effects ($p < 0.001, \chi^2 = 22.658, df = 4$) and 2-way effects ($p < 0.001, \chi^2 = 35.555, df = 6$). Inspection of the resulting partial associations tests showed that, among the effects of relevance to the current study, the interaction between fishing gear and the presence of an injury was of borderline significance (Chi-square = 3.497, p = 0.061, df = 1). Injury rates were estimated to be 11.7% at traps and 6.3% at casitas. Notably, there also was no significant interaction detected between fishing gear, maturity, and the presence of an injury ($p = 0.342, \chi^2 = 0.558, df = 3$). In addition, there was no significant interaction between injury, gear, and season ($\chi^2 = 1.427, p = 0.232, df = 1$). Here
maturity was also treated as a categorical variable with lobsters < 80mm CL considered immature.

Among other interactions of relevance, there was a significant interaction between the treatment of interest (fishing gear: casitas versus traps) and BPI ($\chi^2 = 11.308$, $p = 0.001$, df = 1) but no significant interaction between BPI, gear and season ($\chi^2 = 0.507$, $p = 0.476$, df = 1) or BPI, gear and maturity ($\chi^2 = 1.118$, $p = 0.290$, df = 1). Lobsters at casitas had higher mean BPI (14.5, s.d. = 5.10) than at traps (11.1, s.d. = 3.71).

*Starvation Experiment*

Lesions and necrosis consistent with the appearance of shell disease (see Porter 2004 & Roald *et al.* 1981) were seen on lobsters held in traps (Figure 3). Lesions and necrosis were typically worse on the (tail fan) telson and uropods but also present on pleopods and the abdomen.

A Chi-square analysis showed that there was a significant relationship between the presence of shell disease and starvation time in traps (i.e., weeks held in traps without being fed) in Abaco ($G = 46.644$, $p < 0.005$, df = 2). Further investigation of a contingency table showed that shell disease prevalence increased from 1.4% of the 72 lobsters before placement in traps, to 50% of the 12 lobsters that remained after three weeks in traps, to 70% of the 10 lobsters remaining after 6-12 weeks of captivity in traps. The reductions in number of observations per week were due to deaths and escaped lobsters in early iterations of this experiment when traps were left open. Further exploration (Figure 4) of these data also showed that BPI decreased with starvation time with 90% of lobsters having extremely low values (BPI ≤ 5) by 6-12 weeks.
Figure 3: Shell disease on lobsters held in traps during starvation experiments in Abaco and New Providence, Bahamas. Figures 3a and 3c show a ventral and dorsal view respectively of necrosis on the telson and uropods. Figure 3c also shows two lesions on abdominal segments. Figure 3b shows the normal appearance of the abdomen and tail fan of a lobster.
Lobsters that survived to 9-12 weeks in traps were often heavily fouled by organisms attached to their pleopods, telson, and uropods. This is evidence of decreased grooming or perhaps a compromised immune system. One of the most common fouling organisms appears to be multicellular red algae (Figure 5). Multicellular green algae and multicellular red algae visible to the naked eye also co-occurred on pleopods (Figure 6).
Figure 5: True color microscopic view (x40 magnification) of a section of pleopod of a lobster held in a trap for 12 weeks. The inset shows a further magnified (digitally) and sharpened close-up of multicellular red algae attached to the cuticle. Clumps of other fouling organisms are also seen on setal barbules.
Figure 6: Picture of pleopod clip of a lobster held in a trap for 12 weeks. Red algae are seen on the tan colored portion of the pleopod while green algae are seen on the black portion of the pleopod.

In a second starvation experiment trial conducted in New Providence, 18 of 21 lobsters had shell disease or abrasions at the beginning of the experiment. After three weeks of confinement in traps all lobsters had shell disease. To track the further progression of the disease, percent coverage of the telson and uropods was recorded. Percent coverage was not recorded at the beginning of the experiment. At three weeks the mean percent coverage was 66% (s.d. = 14.0) and by 6 weeks it was 76% (s.d. = 13.5). The mean BPI of these lobsters was 11.0 (s.d. = 2.5) before placement in traps, 9.0 (s.d. = 2.4) after 3 weeks in traps, and 7.6 (s.d. = 1.8), after 6 weeks (Figure 7).

A 1-way repeated measures ANOVA showed that there was a significant difference (F = 56.830; p < 0.005, df = 2) in BPI within subjects due to starvation time. Follow up pairwise comparisons further revealed that each time factor (before, three weeks and six weeks) was significantly different from other time factors (p ≤ 0.007 in each instance).
Figure 7: Box plots of BPI verses time spent in traps without being fed. Upper and lower lines of the box represent upper and lower quartiles whereas the solid line represents the mean. Whiskers show the lowest and highest values.

**Predation experiment**

**Tethering Results**

A logistic regression with lobster carapace length treated as a continuous variable revealed that there was no significant difference in the survival of lobsters based on treatment (i.e., casitas vs. natural shelters; \( p = 0.797 \)), sites (\( p = 0.063 \)), or lobster size (\( p = 0.160 \)). Although the model only explained 24.4% of the variance observed (Nagelkerke \( R^2 = 0.244 \)) it was a good fit for the data as revealed by a Hosmer and Lemeshow significance test (\( p = 0.390, X^2 = 8.460, df = 8 \)).
Still Picture Analyses

From the 51 separate camera deployments I examined 17,659 still photos in which lobster predators were present in 50% (13/26) and 36% (9/25) of the camera deployments at casitas and natural structures, respectively. Among lobster predators, the most abundant was Canthidermis sufflamen (Mitchill 1815) - ocean triggerfish (Table 1; n = 13 at casitas; n = 5 in natural habitat) including 445 appearances in photos at casitas and 10 in natural habitat. There was a greater diversity of lobster predators in natural habitat (n = 4) compared to casitas (n = 1), including: Caretta caretta (Linnaeus 1758) - loggerhead turtle, G. cirratum - nurse shark and Epinephelus striatus (Bloch 1792) - Nassau grouper (Table 1). A mean of 4% of photos per observation / camera deployments (s.d. = 10.19) had lobster predators at casitas, compared to only 0.15% (s.d. = 0.36) in photos taken in natural habitat (Figure 8). Large piscine non-lobster predators were also observed in still pictures near casitas including Lutjanus analis (Cuvier 1828) - mutton snapper, Sphyraena barracuda (Edwards 1771) - great barracuda, and Carcharhinus perezii (Poey 1876) - Caribbean reef shark.

Though more lobster predators were detected near casitas, there was no significant difference in mean percentage of predator occurrence between casitas and natural areas or attributable to an interaction between site and treatment (Table 2). However, there was a significant difference in the occurrence of predators between sites, with more predators appearing in photos taken at Marsh Harbour.
Table 1: Lobster and apex predators detected in still pictures at casitas and in natural lobster habitat

<table>
<thead>
<tr>
<th>Predators detected</th>
<th>CASITAS</th>
<th></th>
<th>NATURAL HABITAT</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>n</td>
<td>% of camera deployments</td>
<td>Total number</td>
<td>% of camera deployments</td>
</tr>
<tr>
<td></td>
<td></td>
<td>with predator</td>
<td>of photos with</td>
<td>with predator</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>predator</td>
<td></td>
</tr>
<tr>
<td>Ocean triggerfish</td>
<td>13</td>
<td>50.0</td>
<td>445</td>
<td>4.52</td>
</tr>
<tr>
<td>(Canthidermis sufflamen)*</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hawksbill turtle</td>
<td>0</td>
<td>0.0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>(Caretta caretta)*</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nurse shark</td>
<td>0</td>
<td>0.0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>(Ginglymostoma cirratum)*</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nassau grouper</td>
<td>0</td>
<td>0.0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>(Epinephelus striatus)*</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mutton snapper</td>
<td>11</td>
<td>42.3</td>
<td>92</td>
<td>0.93</td>
</tr>
<tr>
<td>(Lutjanus analis)*</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Great barracuda</td>
<td>6</td>
<td>23.1</td>
<td>65</td>
<td>0.66</td>
</tr>
<tr>
<td>(Sphyraena barracuda)*</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Caribbean reef shark</td>
<td>1</td>
<td>3.8</td>
<td>1</td>
<td>0.01</td>
</tr>
<tr>
<td>(Carcharhinus perezi)*</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

n-Number of camera deployments in which predator species was detected; Total number of camera deployments: Casitas – 26, Natural habitat-25; Total number of photos taken = 17,659; Approximately 200-500 photos were taken per deployment; *-lobster predator; +-apex predator
Figure 8: Error bar plots of percentage of time-lapse photos in which lobster predators were seen at Sandy Point (SP) and Marsh Harbour (MH) Abaco, Bahamas. Open circles show the mean. Whiskers show 2 standard errors.

Table 2: A Mixed Model 2-way ANOVA testing the effect of Treatment (casitas vs natural areas) on predator occurrence.

<table>
<thead>
<tr>
<th>Source</th>
<th>Sum of Squares</th>
<th>Df</th>
<th>Mean Square</th>
<th>F</th>
<th>Sig.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Treatment (Casita vs. Natural)</td>
<td>995.697</td>
<td>1</td>
<td>995.697</td>
<td>8.034</td>
<td>0.216</td>
</tr>
<tr>
<td>Error</td>
<td>123.935</td>
<td>1</td>
<td>123.935</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Site</td>
<td>1714.165</td>
<td>1</td>
<td>1714.165</td>
<td>14.735</td>
<td>&lt;0.005</td>
</tr>
<tr>
<td>Treatment * Site</td>
<td>123.935</td>
<td>1</td>
<td>123.935</td>
<td>1.065</td>
<td>0.307</td>
</tr>
<tr>
<td>Error</td>
<td>5467.786</td>
<td>47</td>
<td>116.336</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td>42798.000</td>
<td>51</td>
<td>116.336</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Corrected Total</td>
<td>8322.000</td>
<td>50</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

a. R Squared = 0.343 (Adjusted R Squared = 0.301)
DISCUSSION

The current study attempts to parse through a handful of factors surrounding the use of casitas and traps in The Bahamian lobster fishery as pertains to the ecological sustainability of the fishery. No obvious negative ecological impacts on lobsters resulting from the use of casitas were detected. Instead, risks associated with the use of lobster traps were revealed. The factors investigated included predation, growth, and disease rates.

One of the key factors evaluated in the current study is predation at casitas. I did not detect a difference in the survival of lobsters between casitas and nearby natural habitat when the adults and juveniles were jointly considered. There also was no difference in survival between adults and juveniles (defined as lobsters < 80mm CL) that could be attributed to casitas. Similar research in Florida, that also involved the tethering of lobsters, detected lower survival of small juveniles (lobsters < 35 mm CL) at casitas but no difference in the survival of lobsters > 60mm CL Gutzler et al. (2015). This is in contrast to Mexico where the survival of juveniles (46-55 mm CL) was enhanced at casitas while larger juveniles (56-65mm CL) suffered higher predation (Eggleston et al. 1992). Although there may be a true difference in the survival of lobsters between Florida, Mexico and The Bahamas, the differences observed between the studies are also likely influenced by the differences in the size classes studied. In addition, a broader suite of ecological impacts, such as impacts on community structure and lobster distribution also need to be considered (Butler and Herrnkind 1997). This includes the availability of natural shelter. For example, it has been reported that in Florida Bay that casitas increased the survival of juvenile P. argus in locations where natural shelters were limited (Mintz et al. 1994). It has also been demonstrated that local retention and improved survival of juvenile P. argus is possible if appropriately sized and spaced artificial shelters are utilized (Butler and Herrnkind 1997).

There is also a possibility that casita design may account for some of the differences between Florida, The Bahamas and Mexico. Predator size, prey size and predator types all influence survival at artificial structures (Hixon and Beets 1989). For example, predator access may be limited due to the size of a casita’s entrance (Hixon and Beets 1989, Arce et al. 1997) and the ease with which lighter Bahamian casitas can be lifted by the likes of nurse sharks, dolphins, and turtles. Casitas deployed in studies conducted in Florida and Mexico were made of concrete and PVC (Arce et al. 1997, Gutzler et al. 2015) and were sturdier than corrugated metal casitas utilized in The Bahamas. The types of predators present will also be influenced by the
ability of casitas to provide refuge for them (Arce et al. 1997, Gutzler et al. 2015). Although casita design and assessment methods may account for some of the observations noted in each study, actual differences in predator assemblages and the availability of alternate prey options are no doubt also factors. The multiple factors that have been documented or suspected of influencing lobster survival at artificial shelters lends credence to the conclusion of Gutzler et al (2015) that rates of predation on lobsters in casitas may be location specific. Despite this, the variability in predation results between locations is also evidence that casitas do not disproportionately influence predation. Indeed, habitat limitation may also play a large role in survival. Using visual surveys, enhanced survival and increased biomass of juveniles were reported in Mexico at locations where habitat was limiting (Briones-Fourzán et al. 2007). In addition to habitat availability, yet another factor that may have played a role in what was observed in Mexico is enhanced protection of smaller juveniles due to the ability to cohabitate with larger lobsters under casitas (Briones-Fourzán et al. 2007).

Due to limitations with the use of tethering for the measurement of predation (as described in the Methods section), I enhanced my assessment of predation risk for lobsters in casitas and natural shelter with still photos. The results from that additional approach confirmed that there was no clear difference in predation risk between casitas and nearby natural shelters, though a greater diversity of predators was recorded in natural areas. Predation risk assessed by both tethering and videography also revealed differences between my study sites at Sandy Point and Marsh Harbour regardless of treatment (i.e., casitas or natural areas). This difference is probably due not only to differences between sites in predator type, abundance, and size but also to differences in the protection afforded by the available natural shelters. For example, many fewer physical shelters (e.g., holes, sponges, seagrass ledges) exist in the lagoon at Marsh Harbor under which lobsters could be tethered as compared to Sandy Point. Therefore, a number of lobsters at Marsh Harbor were tethered in seagrass. Other than when larvae recruit from a planktonic existence to the sea floor, lobsters do not normally use seagrass as refuge when crevices exist, but dense seagrass offers some degree of refuge (Eggleston et al. 1992) compared to sand for juvenile and adult lobsters. Regardless of the cause, predation risk for lobsters varies not only on regional scales (e.g., Florida, Mexico, Bahamas) but also among locations within a region (Mills et al. 2008) as demonstrated in this study and that by Gutzler et al. (2015).
Another facet to the debate on whether casitas or traps are detrimental to lobster fisheries centers around lobster health. The current study evaluated a number of aspects of lobster health including the dependent variables: the occurrence of injuries, molt stage (a proxy for growth rate), and BPI. Predictor variables included season (open vs. closed), fishing gear (treatment of interest), and lobster maturity.

I found that there was no significant difference in the presence of injuries in lobsters in traps and casitas, though many would argue that the difference between \( p = 0.061 \) and the significance threshold (\( p = 0.05 \)) is inconsequential or that this is perhaps a statistical power issue. However, the non-availability of lobsters from natural locations precluded a comparison to wild lobsters, thus limiting interpretation of these results. It should also be borne in mind that injuries are a common natural occurrence among decapods (Juanes and Smith 1995). The differences between casitas and traps considered here might also be viewed as being of minor consequence when compared to other factors that typically have a major influence on injury rates. For example, injuries that result from handling by fishermen when lobsters are caught and released are likely to be of greater consequence. This occurs at casitas and with traps when fishers release sublegal sized or egg-bearing lobsters. Bycatch issues like this are considered in Chapter III of this dissertation. Nonetheless, injury rates of 6.3% at casitas 11.7% for traps in the current study, are relatively low compared to ~25% reported for the Dry Tortugas, Florida (a non-fished area) or 34.8% for the Virgin Islands National Park (Davis 1981). It is also on the lower end of injury rates reported for a number of *Homarus americanus* (American lobster) fisheries reported in Davis 1981. Managers must bear in mind that fishery induced injuries can result in a marked reduction in the growth rates of smaller *P. argus* (Davis 1981, Hunt and Lyons 1986) or in eventual death in crustaceans (Bergmann and Moore 2001). Another outcome of a reduction in growth rates of juvenile lobsters is the likelihood that it will take longer to attain a legal size (Davis 1981). Although these effects are not of concern for lobsters collected from traps, bycatch including sub-legal size lobsters and lobsters bearing eggs would be affected. This may also affect large lobsters if a maximum size limit is imposed as recommended in *St. George’s Declaration on Conservation, Management and Sustainable Use of the Caribbean Spiny Lobster (Panulirus argus)*. These effects would be also compounded by exposure. Though the severity of effects varies according to the type of injury (example: cheliped vs walking legs,) and the amount of limbs lost by crustaceans, community-and even population-wide effects are
also possible (Juanes and Smith 1995). This is due to the energetic costs, reduced success when foraging and reduced mating success (Juanes and Smith 1995). It should also be borne in mind that an increase in predation resulting from injuries has also been reported for *P. argus* (Parsons and Eggleston 2006). This may also mean that a failure to detect higher injury rates at casitas in the current study might be due to the absence of injured lobsters that were already preyed upon. Notably, in the current study, a logistic regression showed that there were no significant interactions between fishing gear and molt stage, hence, no differences in growth rates were detected that were influenced by fishing gear.

Inspection of the interaction between gear and BPI showed that lobsters in casitas had higher BPI than those in traps, suggesting that lobsters in casitas have eaten more recently and that they are healthier than those in traps. Lobsters are free to enter and leave casitas at will, but this is generally not so for legal-sized or near legal-sized lobsters once they enter a trap. As the results of my starvation study show, the longer a lobster remains constrained in a trap the lower its BPI until it asymptotes at BPI values of 3-5, which are equivalent to readings of just seawater. In short, lobsters that utilize casitas are generally in better condition. If lobsters remain in traps too long, either during the closed season or when left in the traps as social attractants to other lobsters, then this will have negative repercussions for growth and perhaps survival even if they escape or are released. The odds of survival of lobsters that remain in traps are also reduced due to a higher risk of cannibalism brought on by poor nutrition (Jeffs and Davis 2003). Post-molt individuals are at greatest risk (Matthews and Maxwell 2007). Despite the ability to eat conspecifics, mean BPI was lower for surviving lobsters in traps after a few weeks. The poor nutritional condition of lobsters obtained from traps is most likely due to their inability to forage.

Another consideration in the debate on the use of casitas verses traps in the Bahamian commercial lobster fishery is the possibility that one or both types of gear increase the prevalence of diseases. Based on visual inspection for late stage signs of PaV1, none of the 600+ lobsters collected in traps or casitas appeared to be infected with PaV1 at Marsh Harbour or Sandy Point. This is consistent with the results of a study conducted in 2010 that employed a more sensitive PCR test to detect PaV1 in lobsters obtained from the commercial fishery at Sandy Point (Moss et al. 2013). None of the 98 lobsters tested in 2010 were infected. Use of PCR is more accurate as it allows detection of smaller viral loads and early stages of infection (Montgomery-Fullerton et al. 2007) compared to gross visual signs of infection that are an
indication of late stage infection and often absent in infected adults (Behringer et al. 2011). A further consideration is that juvenile lobsters are more susceptible to the virus (Shields and Behringer 2004, Butler et al. 2008, Huchin-Mian et al. 2008, Briones-Fourzán et al. 2009), and my study included many more lobsters as well as sublegal lobsters down to a size of <30 mm CL whereas Moss et al. (2013) only sampled legal-sized lobsters. The absence of PaV1 infected lobsters in Marsh Harbor and Sandy Point based on my results and those of the Moss et. al (2013) are also consistent with reports by Abaco based fishers that lobsters displaying clinical signs of advanced PaV1 infection (i.e., lethargy, milky white hemolymph, heavy fouling of carapace) are not seen in Sandy Point and Marsh Harbour, although they are occasionally seen further north in Abaco. The virus has only been detected at three of eight sites and at low prevalence in other parts of The Bahamas (Moss et al. 2013). Therefore, there is no evidence that PaV1 currently poses a major threat to lobsters and the lobster fishery in The Bahamas. However, periodic monitoring for PaV1 would be prudent because prevalence levels fluctuate in other fisheries (Behringer et al. 2011) and PaV1 has indeed appeared in locations where it was not previously documented (Briones-Fourzán et al. 2009). Yet, because of the absence of PaV1 infections at my study sites, it is not possible to draw any conclusions as to the potential effects of casitas vs. traps on the spread of PaV1. In Florida, the use of lobster traps may facilitate the spread of PaV1 as infection rates were higher for non-infected lobsters when confined in traps with infected lobsters (Behringer et al. 2012). In addition, non-infected lobsters tended to avoid traps with infected conspecifics. Thus lobster catches may be lower due to mortality resulting from disease transmission and due to lobsters avoiding traps. In Mexico, where a casita based P. argus fishery exists, no link between casitas and PaV1 was detected as prevalence did not differ significantly between natural areas and casitas (Huchin-Mian et al. 2013). It is also suggested that this may be influenced by the ability of non-infected lobsters to simply select other casitas as casitas do not confine lobsters unlike traps (Huchin-Mian et al. 2013). There is also a possibility that the addition of casitas to areas where habitat is limited could help to alleviate habitat limitation, thereby reducing crowding and the risk of PaV1 transmission in natural shelters.

Similarly, the prevalence of other diseases in relation to fishing practices is of relevance to the fishery for P. argus in The Bahamas as is the case in other crustacean fisheries worldwide. For example, shell disease has been documented in a number of species of crustaceans (Roald et al. 1981, Porter 2004, Davies et al. 2014). Its occurrence in the American clawed lobster (H.
Americanus) dates back nearly 100 years, whereas its presence in P. argus was not recorded until 1998 (Porter 2004). This highly pathogenic disease emerged as a concern for New England’s H. americanus fishery in 1998, and has subsequently resulted in major economic losses in that region and the call for a closure of the lobster fishery in southern New England (Castro and Angell 2000, Davies et al. 2014). In the current study, lesions consistent with those of shell disease (as described by Porter 2004 and Roald et al. 1981) and fouling organisms (Figures 5 and 6) were only detected in lobsters that were held in traps for several weeks. Every lobster that was held in a trap for at least six weeks developed lesions consistent with shell disease. Signs of shell disease were not observed in lobsters found at casitas. Confinement is a well-known risk factor for shell disease among crustaceans (Roald et al. 1981). In addition, as lobster nutritional condition declines with duration of confinement, they cease grooming, which may facilitate the spread of shell disease and other afflictions such as encrustation with epiphytic algae (Behringer et al. 2008) as documented in the current study. Lobsters may escape affliction with shell disease by molting (McLeese 1965). Although shell disease does not appear to pose a major risk to P. argus fisheries, the use of casitas appears to further minimize risk while traps increase risk. Diseases associated with unnecessarily long periods of confinement in traps, as might occur during the closed season or due to ghost fishing, pose an unnecessary and avoidable risk to the fishery. There are reproductive considerations as well. Reproductively active females normally undertake deep-water migrations to release eggs (Bertelsen 2013). Remaining in traps would stifle this behavior whereas casitas do not confine lobsters.

In summary, the current study has not revealed any obvious negative ecological impacts on lobsters resulting from the use of casitas. Unlike other studies that have shown that small juvenile lobsters (<35 mm CL) are exposed to higher rates of predation at casitas compared to natural habitats (Gutzler et al. 2015), I found no difference in the survival or predation risk of sub-adult or adult lobsters (> 80 mm CL) between casitas and natural shelters. This conclusion held despite the different levels of predation risk detected between Sandy Point and Marsh Harbour. In addition, the current study failed to detect a negative effect of casitas on lobster nutritional condition, growth, or infection by disease. These are key aspects of fishery sustainability. In contrast, the health of lobsters left in traps for more than a few weeks diminished significantly and they were much more susceptible to shell disease and encrustation
by fouling organisms. There is a strong indication that traps have the potential to be destructive to the lobster population and thus their use needs to be regulated and closely monitored.

Whereas the current study explores key aspects of the effects of casitas and traps on lobster biology and ultimately, fishery sustainability, many more influences are of relevance. Consideration of these complex issues by managers is crucial and is further complicated by the need to weigh science along with the socio-economic concerns of stakeholders. Included among ecological factors to be considered are effects of casitas on community structure and the effects of alternate shelter availability (Butler and Herrnkind 1997). This also includes the effects of large numbers of lobsters residing in casitas set in seagrass instead of (or in addition to) natural crevices that are often away from seagrass, as was the case in Marsh Harbour. As it plays a predatory role, it is suspected that a reduction in the numbers of *P. argus* could result in changes in the number and proportion of species it preys on (Davis and Dodrill 1980). By extension, it may be reasonable to assume that excessive numbers of lobsters into seagrass habitats could also alter community structure in that habitat. However, there is no evidence of this impact as yet, in Florida mollusk abundance and diversity were controlled by factors other than the number of lobsters in casitas (Nizinski 2007).

Other far ranging ecological considerations also apply, for example, casitas shade seagrass, which leads to the death of seagrass at casitas and possible effects on primary production depending on the number of casitas. Additional issues of more direct impact include bycatch. In the case of The Bahamas, this includes bycatch associated with casitas and traps. For example, a gaff-like hook is used to retrieve lobsters from casitas in The Bahamas. It is not known whether undersized lobsters survive after being hooked and released by fishermen. Chapter III of this dissertation documents and explores the implications of bycatch in the fishery.
CHAPTER III

SIZE SELECTIVITY AND BYCATCH MORTALITY OF LOBSTERS AND OTHER TAXA IN CASITAS COMPARED TO TRAPS IN THE BAHAMIAN LOBSTER FISHERY

INTRODUCTION

Bycatch has been defined in varying ways (Kirby and Ward 2014) ranging from “catch that is unused or unmanaged” (Davies et al. 2009), to “the incidental take of undesirable size or age classes of the target species (e.g., juveniles or large females), or the incidental take of other non-target species” (Lewison et al. 2004). All of these definitions make clear that bycatch includes non-targeted animals that are caught by fishing gear. Most of the disagreement between definitions centers on what is considered targeted verses non-targeted (Davies et al. 2009), including whether organisms that were not necessarily targeted but are still retained should be considered bycatch. This paper will consider bycatch in its broadest sense, including: all non-target taxa, non-target size classes, and other species that fishers retain but would not catch if they were not fishing for the primary species.

Consideration of bycatch, selectivity, and survival of discards is not a new problem (Hall et al. 2000, Harrington et al. 2005). Since the 1950s, when fisheries management chiefly focused on target species, bycatch of non-target species has gradually emerged as a major issue that fisheries managers are compelled to address (Hall et al. 2000, Davis 2002). In some instances, bycatch amounts exceed those of the target species, which is generally recognized as being unacceptable (Campos and Fonseca 2004). Reducing bycatch mortality is considered a key tenet of fisheries management and is thus specifically mentioned in the United Nations’ globally agreed FAO Code of Conduct for Responsible Fishing (Uhlmann and Broadhurst 2015). Moreover, the minimization of bycatch has repercussions to natural and human systems ranging from altered biodiversity and ecosystem disruption to increased fisher conflicts and operational costs (Crowder and Murawski 1998, Hall et al. 2000, Li et al. 2004, Harrington et al. 2005, Kelleher 2005). Despite a rise in awareness of bycatch by managers over the past six decades, measures of bycatch are still all too often absent or rife with uncertainty (Lewison et al. 2004,
Zeller and Pauly 2005). Amid this uncertainty, global estimates put bycatch levels at 25-40% of worldwide catches of target species (Davis 2002, Davies et al. 2009). With global fisheries production estimated at 62.7 million tons in 2011, bycatch was thus potentially 15.7 – 25 million tons up to that time.

Although many species obtained as bycatch are viewed as having no appreciable economic value, for fishing practices to remain sustainable they must consider impacts not only to target species but also to bycatch species of presumed lesser value whose ecological roles may be significant (Garcia et al. 2012). Indeed, excessive fishery bycatch can have negative repercussions for populations subject to bycatch, to the food webs within which they interact (Crowder and Murawski 1998), and perhaps even to the ecosystems in which they reside (Harrington et al. 2005).

Organisms obtained as bycatch are either discarded uninjured, discarded alive but with injuries, discarded dead, or retained for consumption. Separate estimates for each of these components of bycatch are often lacking. Whereas the fate of retained bycatch or that which is discarded dead is understood (i.e., death), the probability of mortality for discards-injured or not-is usually unknown, thus undermining fisheries stock assessments by overestimating population sizes (Uhlmann and Broadhurst 2015). In addition, efforts to ameliorate bycatch and its impacts may be hampered if its true severity is underestimated (Crowder and Murawski 1998). In terms of fishery economics and production, the discarding of dead or injured bycatch is wasteful and other markets may exist for that bycatch. For example, the tremendous growth in aquaculture experienced globally has led to the realization that there is often insufficient fish meal readily available as food for cultured animals, and bycatch could serve as a major source of feed (Li et al. 2004).

In many parts of the Caribbean and in The Bahamas, casitas (also known as pesqueros or condos) have become popular with fishermen over the past few decades (FAO/WECAFC 2007). Depending on local traditions and regulations, casitas are one of a variety of methods used to capture Caribbean spiny lobster-\textit{P. argus}. Disparities in the amount and character of bycatch potentially exist between the two primary types of fishing gear (i.e., casitas and wooden-lathe traps) that are utilized in the Bahamian lobster fishery, which is the largest lobster fishery in the Caribbean and the most valuable fishery in The Bahamas.
Landings of *P. argus* were valued at over $800 million annually (FAO 2007) with a production of 35,642 metric tons in the Caribbean (FAO 2015). In The Bahamas, 6088 metric tons (converted to whole weight) were landed in 2013, which was valued at $47.7 million. The fishery in The Bahamas was also classified as “fully exploited or stable” up to 2013 (FAO 2015). Across the wider Caribbean the fishery generates foreign exchange, provides direct employment to 33,000 fishers (Chavez 2009), indirect employment to local peoples and has additional but unknown value for subsistence fishing, recreational fishing, and multiplier effects such as tourism (FAO 2007). This widespread species is also of ecological importance as a predator and as prey, although the role that lobsters play in the functioning of the ecosystem and thus provision of ecosystem services is complex and incompletely understood (Butler and Herrnkind 2000, Butler et al. 2006). The Caribbean spiny lobster is located throughout the Caribbean Sea, through Florida, and north to Bermuda; a subspecies is found in abundance as far south as Brazil (Arce and de León 2001). The species widespread distribution is due in large part to the species' 6-8 month long pelagic larval stage (Goldstein et al. 2008) as is the genetic similarity of lobster populations throughout the Caribbean (Silberman et al. 1994, Silberman and Walsh 1994, Naro-Maciel et al. 2011). Biophysical modeling of larval dispersal around the Caribbean indicates that lobster stocks in the Bahamas are largely determined by local recruitment, but also contribute importantly to stocks elsewhere in the Caribbean by serving as an important source of larvae (Kough et al. 2013).

Casitas vary in size and construction but are usually flat, rectangular structures ~4 m² in area with at least two open sides <10 cm in height that mimic the natural crevice shelters used by lobsters. Thus, casitas do not restrict the movement of lobsters, invertebrates, or fish that use the structures as shelter. However, the method by which lobsters are extracted from beneath each casita by free-diving or fishers using hooks potentially causes a bycatch problem. In the Bahamas, aided by air compressors, fishers typically utilize a gaff-like device locally referred to as a “lobster hook” (Figure 9) to retrieve the lobsters from under the casita. In doing so, lobsters are impaled by the hook shaped end of the lobster hook. If of legal size, the lobsters are retained. But at times, fishers mistakenly hook lobsters that are below the 3 1/4 in (83 mm) carapace length (CL) minimum size limit. Fishery regulations require that undersized lobsters be released, but it is not known whether lobsters impaled by the lobster hook (a roughly 2cm deep puncture wound) and subsequently released will survive. One focus of this dissertation chapter is
determining whether lobsters that are hooked and released survive, and whether the type of injury influences their probability of survival.

Figure 9: Lobster hook utilized in Bahamian lobster fishery. The red rectangle depicts the hook shaped end of a lobster hook that is used to retrieve lobsters from beneath or scurrying from casitas

There are also concerns about bycatch associated with the use of lobster traps used by fishermen, along with the potential for negative ecological effects associated with trap-based fishing (Chiappone et al. 2002). For example, the use of traps may result in damage to sessile taxa such as sponges and corals (Chiappone et al. 2002), habitat damage by derelict traps (Arthur et al. 2014), damage from wind driven movement even when traps are not lost (Lewis et al. 2009), and from benthic disturbance resulting from the repeated setting and retrieving of traps. But perhaps the most troublesome aspect of trap fisheries for spiny lobsters are the effects due to "ghost fishing" by lost traps. When lost, traps continue to attract target and non-target species as shelter, many of which then perish (Bilkovic et al. 2012), including some species of conservation importance (Anderson and Alford 2014). Ghost fishing effects are exacerbated by the length of
time a trap continues to fish, the location where a trap is lost, and the behavior of the species involved (Smolowitz et al. 1978). This is especially true when trap selectivity is broad and trap materials are durable (Johnson 2010). For example, a recent study in Florida where lost wooden lathe lobster traps, of the same design as lobster traps utilized in the Bahamas, continued to fish for a median of 509 days (Butler and Matthews 2015). In addition, differences in trap degradation varied as much as 300+ days depending on the location, (Butler and Matthews 2015).

Though estimates of trap losses range from 10-70% on a global scale (Bilkovic et al. 2012), there is a general lack of information on rates of trap loss (Arthur et al. 2014). Yet another consideration is that the retention of non-targeted size classes of lobster (i.e., sublegal or undersized lobsters) results in reduced future revenues and reduced reproduction of lobsters (Everson et al. 1992). Detection and quantification of the ecological repercussions of trap use in the Bahamas’ main commercial fishery is important and will assist managers with mitigation of these effects where needed. Remedies for some of the disadvantages of traps, such their broad size selectivity and continued lethality if lost, are known. Although traps are considered a primitive fishing gear (Stewart 2007), their designs have changed for the better over time. For example in some fisheries the use of escape vents (Everson et al. 1992) and biodegradable panels permit the escape of bycatch, including sub-legal lobsters (Treble et al. 1998, Arana et al. 2011).

Size selectivity of traps and casitas also has implications for stock assessment where size frequency or age-based assessment methods are utilized. Despite the shortcomings that may result from using chiefly fishery dependent data and the use of size-rather than age-based methods for decapod fisheries for which direct aging methods are not possible, these are often the only data available to some managers (Punt et al. 2013).

Fishery-dependent data generally assume that the age structure of wild populations can be determined from the size-or age-frequency of the organisms caught by fishers. It is thus important to understand how representative the fishery-dependent catches are of wild populations, so biases can be quantified and potentially meliorated. This is especially so when size frequency data are utilized to estimate key parameters such as fishing mortality and biomass (Punt et al.). A preference for the inclusion of size frequency data in analyses is also developing among modelers as this type of data enables one to garner more information about a fishery even when other data that are normally considered ideal are available (Punt et al. 2013). Fishery-
dependent size frequency based or size frequency supplemented lobster stock assessments have been attempted or conducted for The Bahamas and will continue for the foreseeable future (CRFM 2011). Although it is recognized that lobster size-frequencies determined from fishery catch in The Bahamas may be biased, the extent of the biases is unknown.

In this dissertation chapter, I attempt to shed light on the respective bycatch issues and size-selectivity biases of casitas and traps used in the commercial fishery for the Caribbean spiny lobster in The Bahamas. These questions are addressed using collections of lobsters from casitas and traps of commercial fishermen as well as from gear that I deployed, along with experimental studies of bycatch effects on lobster growth, mortality, and disease.

Three studies were conducted to document and ascertain the differences in bycatch and size selectivity of casitas and traps. These include: (a) an experiment on the effect of hooking on lobster survival, (b) a study of lobster trap bycatch, and (c) a comparison of size frequency structure in casitas and traps.

METHODS

Hooking Mortality Experiment

I hypothesized that juvenile lobsters collected in casitas experience increased mortality due to injuries associated with being hooked by fishermen and released. This experiment was conducted to mimic the way in which fishermen use hooks to retrieve lobsters from beneath casitas and to ascertain whether there is a difference in survival of lobsters that were hooked in the abdomen or hooked in the cephalothorax, as compared to a "control" group that had not been hooked. Sixty two (62) lobsters collected by divers in The Bahamas (Orange Cay, Marsh Harbour and New Providence) and an additional 29 lobsters collected by divers in the Florida Keys (USA) (a total of 91 lobsters overall) were included in this study and assigned, in equal numbers, to three treatment groups in experiments conducted in The Bahamas (September 2012, August 2014, November 2015) and Florida (July-August 2015). These included the non-hooked (control) group, a group hooked in the cephalothorax, and a group hooked in the abdomen. In creating the treatment groups in which lobsters were hooked in the abdomen or cephalothorax, I attempted to recreate the types of injuries inflicted on lobsters during commercial casita fishing operations. This involved using a standard lobster hook to create a single, randomly located (within the confines of the abdomen or cephalothorax treatment) hooking injury on the ventral
surface of each lobster. Lobsters were then randomly assigned to wooden lobster traps or to a wire cage depending on the iteration of the experiment. Lobster survival was then checked daily in situ by snorkelers for 10 consecutive days. Dead lobsters were removed from the trap or cage daily. A subset was retained in the traps or cages for 15 days and 25 days to ascertain whether the patterns of lobster mortality among groups differed over a longer time frame. The size range of lobsters included in the study was ≥60 mm CL but smaller than the 83 mm CL (3 ¼ in) Bahamian minimum legal size limit, as it is that size range in which fishers may have difficulty in discerning that a lobster is below the legal limit, thus contributing to sublegal size bycatch mortality. The effect of hooking injury on lobster survival after 10 days was tested using 2 x 3 (mortality x hooking treatment) log-likelihood contingency table analysis; a log-linear analysis was used to test mortality after 10, 15 and 25 days.

**Trap Bycatch**

I further hypothesized that traps have a higher bycatch of other taxa compared to casitas, but minimal effect on undersized lobsters. Other than the inadvertent injury of under-sized lobsters due to hooking there is no bycatch of other taxa associated with casitas, so I focused my evaluation of non-lobster bycatch on traps. Therefore, I obtained estimates of the weight of bycatch and the weight of lobsters caught in wooden slat traps by commercial fishers using a kitchen balance and a spring balance. To this end, an observer was placed aboard a commercial lobster trap fishing boat on three occasions and at three locations: Cay Lobos (March 2013; n = 124 traps), South Andros (May 2013; n = 392 traps) and a bank area southwest of the Tongue of the Ocean (March 2015; n = 50 traps) (Figure 1, Chapter II). Differing sampling strategies were used by the fishers and their success rates (i.e., catch per trap and ability to sample catch) varied among the trips, resulting in tradeoffs in the type and quantity of data that I collected.

Estimates of the total bycatch were obtained for each trip. At Cay Lobos observers noted the bycatch from all traps, whereas at South Andros roughly half of the traps were sampled. This included noting the proportion of traps that contained bycatch, the weight of lobsters retained, the species of bycatch, and the fate of the by-caught species (i.e., retained, discarded dead, or discarded alive). Not all bycatch was measured due to the fast pace with which fishers worked each trap. The sampling trip near the southwest region of the Tongue of the Ocean (SWTO) utilized a different sampling strategy. The observer randomly selected one trap in each string of
25 traps and noted the entire contents of the selected traps including the number and weight of lobsters, as well as numbers and weight of bycatch species. A total of 50 traps were sampled in this manner in the SWTO region where 1250 traps were actually fished. Weights of bycatch species and lobsters were successfully obtained for two of the three trips including South Andros and SWTO, permitting estimation of bycatch as a proportion by weight of the target species to be estimated for the two locations. In the case of South Andros, the total weight of lobster tails for the commercial fishing trip (91lbs / 41.28 Kg) was converted to whole weight using a factor of 3 (Muller et al. 2000). At SWTO, lobster weights were already recorded as whole weights. However, the total bycatch for the trip was estimated by multiplying by 25 as 1/25 of traps were sampled. Because half of the traps were sampled at Cay Lobos, bycatch estimates were doubled to estimate the entire catch for the trip. Numbers were used at Cay Lobos instead of weight due to a malfunctioning scale. Weights are, nonetheless, presented for SWTO and South Andros.

Weights were not collected for three species including the sea cucumber *Holothuria Mexicana* (Ludwig Diels, 1875) the crab *Dromia erythropus* (George Edwards, 1771), and an unknown gastropod. In some instances, for example the hermit crab *Dardanus venosus* (Milne Edwards 1848), a small portion of specimens were not weighed. In these instances the mean weight was used to estimate total weight.

*Size-Selectivity of Lobsters in Traps and Casitas*

Finally, I hypothesized that wooden-lathe traps and casitas differed in the size frequency of lobsters associated with each gear and in the number of legal sized lobsters that each gear would retain. During the summer of 2012, I deployed experimental casitas (n = 6) and traps (n = 10) in 3-5m depth water at each of two locations off Abaco Island in The Bahamas (Sandy Point and Marsh Harbour; Figure 1, Chapter II), within fishing grounds where commercial fishing takes place. The locations selected were 1-3 km from shore and were representative of the shallow shelf areas (Sandy Point) and shallow back reef lagoon areas (Marsh Harbour) that typify commercial lobster fishing grounds in The Bahamas. Fisher casitas and traps were also sampled to ensure that lobsters sampled were representative of the commercial fishery. Lobsters were collected from casitas by SCUBA divers or free divers using nets and tail-snares, whereas traps were pulled aboard a vessel for sampling.
Commercial casitas are left on the seafloor during the closed season and lobsters are found under casitas year round. Therefore, my sampling took place during both the closed season (April 1 – July 31) and open season to ensure that the lobsters sampled were representative of all lobsters that would be associated with casitas. In particular, there are reproduction related seasonal migrations related to mating and egg release that may result in seasonal differences in lobster size frequencies at fishing gear locations. A further consideration was that fishers remove legal sized lobsters when the fishery is open. Thus, these lobsters would be underrepresented, resulting in biased sampling if sampling were limited to the open season.

The casitas and traps were built to the specifications of typical commercial lobster fishing gear. Casitas in the fishery vary somewhat in size and construction materials used, but most are approximately 1.8m x 1.2m with a ceiling made of corrugated metal and legs composed of 1.8m lengths of 5cm x 13cm lumber. In addition, both of the 1.2m ends of the casitas are open and permit free movement of lobsters and other taxa in and out of the casita. Consistent with commercial lobster trap requirements outlined in The Bahamas’ Fisheries Resources (Jurisdiction and Conservation) Regulations (1986), experimental traps were 0.9m long x 0.6m wide x 0.6m tall with wooden slats no less than 2.5 cm apart.

At both study sites (Sandy Point, Marsh Harbour), experimental casitas and traps were placed in seagrass, as is the practice of fishers. Casitas were generally placed approximately 90m apart in a single line, whereas traps were placed in a line approximately 300m apart. These distances were chosen to balance the need to ensure that there was minimal to no interaction between gear types with the need to ensure differing environmental conditions did not affect results. Traps were deployed for 2-21 days, to mimic the length of time fishers leave traps in the water during the open season. Fishers also report that traps must remain in the water (become “seasoned”) for a few weeks before they attract lobsters, so I did so as well.

The carapace length of all lobsters found under casitas and retained in traps was measured, and a two-sample t-test used to compare the mean size of lobsters found at each gear type. The t-test was conducted on ranked carapace lengths due to violation of normality and equality variance assumptions. Size frequency distributions were also compared using a number of population parameters (e.g. range, skewness, kurtosis, standard deviation) and visual inspection of size frequency histograms.
RESULTS

*Hooking Mortality Experiment*

Of the 91 lobsters that were used for this experiment, 16 (17.6%) died within 10 days. Of those that died, seven were hooked in the cephalothorax, five in the abdomen (“tail”) and four were not hooked (Table 3). The mean number of days of survival were 0.8, 2 and 1.3 for lobsters hooked in the cephalothorax, abdomen, and not-hooked respectively among lobsters that died. A Kaplan-Meier survival analysis showed that there was no significant difference in the survival of lobsters within ten days (Tarone-Ware statistic = 0.918, df = 2, p = 0.632). Examination of a cumulative survival curve showed that mortalities were immediate (within 1 day) in some instances and that all mortalities occurred by the seventh day (Figure 10).

<table>
<thead>
<tr>
<th>Injury Site</th>
<th>Number of Lobsters that Died</th>
<th>*Mean Number of Days Until Death</th>
<th>Number of Lobsters Sampled (n)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Thorax</td>
<td>7</td>
<td>0.8</td>
<td>31</td>
</tr>
<tr>
<td>Abdomen</td>
<td>5</td>
<td>2</td>
<td>30</td>
</tr>
<tr>
<td>Not-hooked</td>
<td>4</td>
<td>1.3</td>
<td>30</td>
</tr>
</tbody>
</table>

*mean of the lobsters that died

A subset of the lobsters (n = 63) was monitored for 25 days to ascertain whether there was a difference in survival over a longer time period after hooking. After 15 days, one more thorax-hooked and one more abdomen-hooked lobster died, and after 25 days an additional abdomen-hooked and a non-hooked lobster died. Therefore, during the 25 day long trial, five of 63 lobsters died: 5% of the non-hooked lobsters, 14% of the tail-hooked lobsters, and 5% of the lobsters hooked in the thorax. A log-linear analysis examining the independence of injury treatment and the survival of lobsters after 10 days, 15 days, and 25 days survival revealed that none of the relevant two-way effects were significant (Table 4). Thus, the location of the hooking injury on lobsters had no effect on survival and 10 days of observations was an adequate duration for this experiment.
Figure 10: Survival of *P. argus* subsequent to a hooking injury. The survival of lobsters hooked in the abdomen, hooked in the thorax or not hooked (control) are compared. The vertical portions of each line depict the occurrence and magnitude of mortality events. All mortalities occurred by the onset of the seventh day and included lobsters from each hooking injury category.

Table 4: Log-linear Analysis Partial Chi-Square Associations for Injury Location vs. Days of Survival

<table>
<thead>
<tr>
<th>Effect</th>
<th>Df</th>
<th>Partial Chi-Square</th>
<th>Sig.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Injury Treatment x 10-day survival</td>
<td>2</td>
<td>0.057</td>
<td>0.972</td>
</tr>
<tr>
<td>Injury Treatment x 15-day survival</td>
<td>2</td>
<td>2.244</td>
<td>0.326</td>
</tr>
<tr>
<td>Injury Treatment x 25-day survival</td>
<td>1</td>
<td>1.012</td>
<td>0.314</td>
</tr>
</tbody>
</table>

*Trap Bycatch*

Among the three sites sampled, an overall mean of 46% of the lobster traps contained bycatch of other species (Table 5). As one would expect, bycatch varied appreciably among
fishing locations (24-88%). At the South Andros and SWTO locations, bycatch computed as a percentage of lobster weight were 32% and 47%, respectively (Table 6).

<table>
<thead>
<tr>
<th>Location</th>
<th>Number of Traps Sampled</th>
<th>Number with Bycatch</th>
<th>Percent with Bycatch</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cay Lobos</td>
<td>119</td>
<td>29</td>
<td>24.3</td>
</tr>
<tr>
<td>South Andros</td>
<td>392</td>
<td>99</td>
<td>25.3</td>
</tr>
<tr>
<td>*SWTO</td>
<td>50</td>
<td>44</td>
<td>88.0</td>
</tr>
<tr>
<td>Total</td>
<td>561</td>
<td>172</td>
<td>45.9</td>
</tr>
</tbody>
</table>

*SWTO-Near SW Tongue of the Ocean

<table>
<thead>
<tr>
<th>Fishing Ground</th>
<th>Lobster Weight (Kg)</th>
<th>Total Bycatch</th>
<th>Retained Bycatch</th>
<th>Bycatch Discarded Dead</th>
<th>Bycatch Discarded Alive</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Weight (Kg)</td>
<td>Weight (Kg)</td>
<td>Weight (Kg)</td>
<td>Weight (Kg)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>%*</td>
<td>%*</td>
<td>%*</td>
<td>%*</td>
</tr>
<tr>
<td>South Andros</td>
<td>123.8</td>
<td>39.8</td>
<td>32.1</td>
<td>27.9</td>
<td>22.5 [70]</td>
</tr>
<tr>
<td>Near SW Tongue of the Ocean</td>
<td>4682.5</td>
<td>2200.2</td>
<td>47.0</td>
<td>691.8</td>
<td>14.8 [31]</td>
</tr>
</tbody>
</table>

*Percentage calculated as a proportion of the total weight of lobster catch; [ ]-Percentage calculated as a proportion of the total weight of bycatch; Weights at SW Tongue of the Ocean were estimated based on the sampling of 50 traps (1 trap per string of 25 traps) among 1250 traps that were in use by a commercial fishing vessel. Estimates at South Andros were based on the sampling of 392 traps among approximately 800 traps for a commercial vessel.

Total bycatch weight was below 50% overall with bycatch comprising 32% and 47% of lobster weight at South Andros and SWTO (Table 6). Retained bycatch was less than 25% at both sites and dead bycatch that was discarded represented less than 2% of the total weight of the catch. Live discards differed among sites with 67% of bycatch discarded alive at SWTO and
24% at South Andros. Combined, live and dead bycatch discards were 9.5% and 32.2% at South Andros and SWTO, respectively.

Bycatch that were retained by fishers included fishes \textit{[Haemulon album]} (Georges Cuvier, 1830) - white margate, \textit{Pterois volitans} (Carl Linnaeus 1758) - red lionfish, \textit{Epinephelus striatus} (Bloch, 1792) - Nassau grouper, \textit{C. sufflamen} - Ocean triggerfish, \textit{Sparisoma chrysopterus} (Bloch & Schneider, 1801) - redtail parrotfish, \textit{Sparisoma aurofrenatum} (Valenciennes, 1840) - redband parrotfish, \textit{Chaetodon ocellatus} (Bloch, 1787) - spotfin butterflyfish, \textit{Lachnolaimus maximus} (Walbaum, 1792) – hogfish, \textit{Pristigenys alta} (Gill, 1862) - short bigeye], crustaceans \textit{[Mithrax spinosissimus]} (Lamarck, 1818) - West Indian spider crab, \textit{Dardanus venosus} (Milne Edwards, 1848) - starry-eyed crab, \textit{Menippe mercenaria} (Say 1818) – stone crab <claws only>], and molluscs \textit{[Fasciolaria tulipa]} (Linnaeus, 1758) - true tulip. Not all specimens of each species were retained. Bycatch discarded dead by fishers included: \textit{Acanthostracion polygonius} (Poey, 1876) - honeycomb cowfish, \textit{P. volitans}, and \textit{C. ocellatus}, but again, not all specimens of these species were discarded dead.

\textit{Bycatch in Numbers}

In all, 31 species were present in the bycatch of lobster traps deployed by fishers at the South Andros, Cay Lobos, and SWTO locations. The largest fraction of that bycatch were hermit crabs (\textit{D. venosus}; 40% of total bycatch; an estimated 2000 individuals) and lionfish (\textit{P. volitans}; 9% of total bycatch; an estimated 425 individuals), with other crab species (e.g., \textit{M. mercenaria}, \textit{Callinectes sp.}, \textit{Portunus spinimanus} (Latreille 1819) - blotched swimming crab) comprising another major fraction of the bycatch when their numbers are combined (25% of total bycatch; an estimated 102 individuals; Figure 11).

The number of species present in the bycatch at each location varied. There were 18 species detected in the bycatch at each of two sites (SWTO and South Andros), but only 9 species present at Cay Lobos. Dominant species at SWTO included \textit{D. venosus} (61% by number), the invasive lionfish (\textit{P. volitans}; 13% by number), and two common fish: \textit{H. album} and \textit{P. alta} at 4% each. At Cay Lobos, \textit{H. album} (62%) and \textit{P. volitans} (7%) again formed a large component of the bycatch in addition to the triggerfish, \textit{C. sufflamen} (7%). At South Andros, three species of crabs combined to form the most abundant type of bycatch (37%), including a species not previously documented in The Bahamas (Appendix I). The fish \textit{P.}
volitans (15%), H. album (14%) and A. polygonius (12%) were the most abundant individual species at South Andros.

Figure 11: Proportion of Bycatch by Species based on Numbers at three locations: South Andros, Cay Lobos and SWTO

**Bycatch by Weight**

Based on the estimated weight of bycatch, the invasive lionfish *P. volitans* made up the largest component of bycatch at 39%, followed by *D. venosus* (25%), *H. album* (10%) *P. alta* (4%), and *A. polygonius* (4%) at SWTO and South Andros (Figure 12). Cay Lobos was excluded.
from this summary because weights were not collected at this location. Cay Lobos was the only location where the commercially sought Nassau grouper (*E. striatus*) was recorded, but only two specimens were captured in the bycatch.

*Three species of crabs

Figure 12: Proportion of Bycatch by Species Based on Weight near SW Tongue of The Ocean and South Andros
Lobster Size Selectivity Analysis

At casitas, a mean of 10.1 lobsters (n = 46 casitas) were caught per sampling event, whereas a mean of 5.3 lobsters (n = 31) were retained per trap. These estimates exclude the many incidents when no lobsters were present in individual traps (85% of samples) or casitas (50% of samples). This resulted in disproportionate sampling of lobsters associated with each gear type and had consequent repercussions on the statistical tests I used.

The mean size of lobsters in casitas (mean = 71.0 mm CL, sd = 15.3; n = 463) was 10% smaller (t = 8.031, df = 353.7; p < 0.005, Figure 13; Table 7) than the mean size in traps (mean = 79.3mm CL, sd = 10.6, n = 163). Among lobsters associated with casitas, 82% were below the Bahamian minimum size limit of 82.55 mm CL (Figure 13), as compared to traps in which 67% of the lobsters were smaller than the legal size limit. Traps were thus more size selective (range = 88 mm; $s^2 = 112.95$) than casitas (range = 99 mm, $s^2 = 233.15$), resulting in a negatively skewed (g = -0.392, p < 0.01; Table 7), leptokurtic (g = 3.994) size distribution of lobsters in traps. The size distribution of lobsters collected from casitas was positively skewed (g = 0.258) and platykurtic (-1.039) (Table 7; Figure 13).

Among legal sized lobsters (lobsters ≥ 83mm CL) the catch per successful sampling event was 1.78 and 1.74 for casitas and traps (respectively). There was no significant difference in mean size between gear types among this subset of lobsters (p = 0.570, t = 114.419, d.f. = 114.419).
Figure 13: Size frequency distribution (expressed as % of total) of lobsters collected in casitas (n = 46 casitas; 463 lobsters) and traps (n = 31 traps; 163 lobsters).
Table 7: Size Frequency Parameters for Lobsters associated with Casitas and Traps

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Casitas</th>
<th>Traps</th>
</tr>
</thead>
<tbody>
<tr>
<td>N*</td>
<td>46</td>
<td>31</td>
</tr>
<tr>
<td>N’</td>
<td>463</td>
<td>163</td>
</tr>
<tr>
<td>Mean CL</td>
<td>71.0 mm</td>
<td>79.3 mm</td>
</tr>
<tr>
<td>Standard Deviation</td>
<td>15.27 mm</td>
<td>10.63 mm</td>
</tr>
<tr>
<td>Median</td>
<td>71.8</td>
<td>78.9</td>
</tr>
<tr>
<td>Variance</td>
<td>233.15</td>
<td>112.948</td>
</tr>
<tr>
<td>Range</td>
<td>99 mm</td>
<td>88 mm</td>
</tr>
<tr>
<td>Skewness</td>
<td>-0.258</td>
<td>-0.392</td>
</tr>
<tr>
<td>Std. error of skewness</td>
<td>0.113</td>
<td>0.190</td>
</tr>
<tr>
<td>Skewness Critical value</td>
<td>2.28 (p&lt;0.01)</td>
<td>2.06 (p&lt;0.01)</td>
</tr>
<tr>
<td>Kurtosis</td>
<td>1.039</td>
<td>3.994</td>
</tr>
<tr>
<td>Std. error of Kurtosis</td>
<td>0.226</td>
<td>0.378</td>
</tr>
<tr>
<td>Kurtosis critical value</td>
<td>4.60 (p&lt;0.01)</td>
<td>10.57 (p&lt;0.01)</td>
</tr>
</tbody>
</table>

N*-Number of gear with lobsters
N’-Number of lobsters

DISCUSSION

Consistent with what has been observed globally, not enough is known about the amount and composition of the commercial fishing bycatch (Lewison et al. 2004) in The Bahamas. The retention of bycatch by fishing gear is influenced by many things, including, but not limited to: environmental factors, the behavior of the species concerned, fisher behavior, fishing regulations, and fishing gear selectivity (Witherell and Pautzke 1997, Klaer and Polacheck 1998, Belda and Sánchez 2001). The selectivity of fishing gear is particularly important because its ramifications also extend to the accuracy and usefulness of stock assessments that are based on fishery-dependent data and the use of size composition data in an integrated stock assessment approach (Ailloud et al. 2015). This study examined the selectivity of two types of fishing gear-casitas and wooden slat traps-commonly used in the Bahamian lobster fishery, with a focus on
the bycatch of sublegal sized lobster, bycatch of non-target taxa, and size selectivity of the catch of lobsters.

The interaction of fishing gear with target species can lead to undetected mortality due to injuries (Parsons and Eggleston 2007). An example of this from a trap based lobster fishery was seen in *P. argus* in Florida where it has been estimated that there were fishery losses of 160t valued at $1.6 million due to recreational fishery related injuries (Parsons and Eggleston 2007). In the case of casitas, one might assume that bycatch of sub-legal lobsters at casitas is non-existent as lobsters are not restrained from leaving casitas. However, because casitas are fished with the aid of a lobster hook, it is possible that unintended mortality of mistakenly hooked and released sublegal lobsters exists. This is similar to the mortality observed from the release of sublegal sized finfish landed by recreational fishers or catch and release angling (Aalbers et al. 2004). Like fish, death is likely linked to damage to vital organs (Lyle et al. 2007). This kind of undetected mortality (or “cryptic mortality”) of sub-legal individuals (Coggins et al. 2007) has implications for fishery sustainability that needs to be considered by stock assessment modelers (Parsons and Eggleston 2007). Also unconsidered in this study, given the fishery independent nature of the study design, is the bycatch of non-targeted species that are retained by fishers of casitas. Casitas attract many other species besides lobsters and anecdotal evidence indicates that those that are edible are often retained by fishers.

Predictions of sustainable harvest levels rely heavily on mortality estimates and are invalidated when large sources of mortality go undetected. The fact that a large percentage of lobsters (82% of lobsters from casitas and 67% from traps) in the current study were of sub-legal size is evidence that sub-lethal injuries or delayed mortality may be a major problem. The potential for fishers to encounter and interact with sub-legal lobsters is high in casitas where they are especially abundant. Sub-legal lobsters retained in traps, though perhaps fewer than in casitas as my data indicate, may also be subject to cryptic mortality if their confinement is long as occurs in lost "ghost" traps, or in fisheries where it is permissible to use sublegal lobsters as social attractants (e.g., Florida; (Hunt et al. 1986). Although fishers can become good judges of whether lobsters are of legal size, their rapid culling of lobsters from casitas no doubt leads to the unintended hooking of sub-legal lobsters.

Once hooked, fishers have another opportunity to scrutinize the size of each lobster before releasing it, or killing it with a second impalement on the ventral surface of the
cephalothorax before transfer to their vessel. Fortunately, the results of my experiment on hooking mortality indicate that there is no significant difference in the survival of lobsters that were hooked in the abdomen, hooked in the cephalothorax, or not hooked at all. As such, if lobsters are released before receiving a second impalement or released aboard a boat after further scrutiny of their size, a single hooking injury appears to have no effect on mortality for up to a month after hooking. Further, unlike many fishes, the absence of an air bladder in lobsters decreases their susceptibility to barotrauma (Davis 2002) when sublegal sized lobsters are brought to the surface but then discarded alive. Whether the hooking and release of sublegal lobsters reduces their growth or renders them more susceptible to disease was not investigated. Transmission of PaV1 via a contaminated hook is also not considered a current risk as PaV1 was not previously detected in Abaco (Moss et al. 2013) and no lobsters showed signs of PaV1 in the current study.

Exposure and confinement (i.e., handling) can also result in the mortality of sub-legal sized *P. argus* (Matthews 2001). Of the 91 lobsters that I monitored for 10 days, 16 died (17.6%) regardless of the hooking treatment to which they were subjected. By comparison, in Florida 26% of lobsters died after six weeks when sub-legal lobsters were exposed to air for varying lengths of time followed by confinement in traps (Hunt et al. 1986). Whether the incidence of handling mortality that I observed are indicative of that in the casita fishery is unknown because my experimental handling protocol differed from that of fishers who use hooks to retrieve sublegal lobsters from casitas before their release. In particular, I had to confine lobsters in traps after hooking to monitor their recovery. So there is a chance that confinement after hooking increased lobster mortality as compared to lobsters that are released into the wild after hooking. Thus, the mortality observed in the current study are likely higher than what be observed in the fishery.

Given the large proportion of sub-legal lobsters present in casitas and, to a lesser extent, in traps, the survival of these lobsters depends greatly on fisher behavior. Firstly, a fisher must be motivated enough not to target nor retain undersized lobsters. Fisher compliance with the law is influenced by a number of factors including financial tradeoffs, ethical values, and fisher acceptance of the legitimacy of regulations (Hatcher et al. 2000). Second is the ability of fishers to quickly discern sublegal from legal lobsters while working underwater. This would help to avoid unnecessary handling and possible injury to sublegal lobsters. Although my results
indicate that the type of injuries most associated with hooking by fishers do not influence the short-term survival of sublegal lobsters, others have demonstrated the generally deleterious effects of fishery-induced handling and injuries on lobster survival (Hunt et al. 1996, Matthews 2001, Parsons and Eggleston 2007).

The degree to which management is influenced by decisions of fishers also varies depending on the gear used and its selectivity. For example, the results of this study indicate that traps retain a smaller proportion of sublegal lobsters and largely those closer to the minimum size limit in comparison to casitas, which provide refuge to a greater range of lobster sizes (Figure 13). Therefore, if fishers do not tend to comply with minimum size limit regulation, then a harvest strategy promoting the use of traps instead of casitas might be a better management option because fishers would encounter and thus harvest fewer undersized lobsters. On the other hand, if fishers generally avoid hooking undersized lobsters in casitas, then casitas would be the best option because non-targeted lobsters would be subject to less handling. Nonetheless, with traps, selectivity can be further augmented with the use of escape vents (Smolowitz et al. 1978, Miller 1990). A lot is known about the effectiveness of escape vents and their repercussions for the size structure of lobster catches (Nulk 1978, Smolowitz et al. 1978). Managers can select a vent size that allows a more ideal proportion of undersized lobsters to escape while retaining an adequate proportion of legal-sized lobsters (Nulk 1978). All the same, selection of the appropriate gear by managers goes beyond the narrow consideration of lobster size frequency selection and the use of hooks by fishers.

As documented in Chapter II of this dissertation, there is the potential for differential rates of predation on lobsters aggregated in casitas versus those caught in traps. In addition, the hauling of traps can cause physical damage to the benthic environment (e.g., to corals, sponges and seagrass; (Uhrin 2016), and may impose sub-lethal effects on lobsters constrained in traps. There is an equally great or greater concern when traps are lost resulting in "ghost fishing" of lobsters for hundreds of days as observed in Florida (Butler and Matthews 2015). Another misgiving about traps is that even if lost traps are broken up by storms or degrade over time and so do not remain intact to pose a risk of ghost fishing, they nonetheless introduce debris into the marine environment. Concern about fishing gear debris also extends to casitas because they are also susceptible to movement and decay.
Another consideration in the management of commercial fishing gear is the bycatch of non-targeted taxa (Uhlmann and Broadhurst 2015). In The Bahamas lobster fishery, a comparison of the bycatch of non-target taxa would favor casitas whose bycatch is believed to be negligible compared to traps, but still subject to compliance by fishers. I recorded up to 31 species as bycatch in the Bahamian lobster trap fishery, comprising 32%-47% of lobster landings by weight (Table 6). As such, total bycatch estimates from the trap fishery in the Bahamas lay at the higher end of global bycatch estimates. Over an 11-year period (1992-2003), global discards in all fisheries were estimated at 8% (Kelleher 2005). Other estimates of worldwide bycatch place levels between 25% (Davis 2002) and 40% (Davies et al. 2009) of the catches of target species.

When compared to the bycatch associated with Florida's commercial wooden lathe trap fishery for *P. argus*, bycatch in The Bahamas (n = 31 species) is roughly 1/3 of that detected in Florida (n = 89 species; (Matthews et al. 2005). However, it is likely that this difference may simply be due to differences in sample sizes. Approximately 350 traps were sampled in The Bahamas compared to >21,000 in Florida. Among the 89 species detected in Florida, 46 occurred in 0.1% (1/1000 traps on average) of the sampled traps, thus making it likely many would not have been detected with a smaller sample size.

It is also advantageous to consider other measures of bycatch (Vázquez-Rowe et al. 2012), including the species’ exploitation or extinction status. A local species assessment was available for only one species (*E. striatus*) identified as trap bycatch. Although local assessments have not been conducted for the other taxa I recorded as bycatch, global assessments through the IUCN Redlist are available. Consistency between local assessments and IUCN Redlists have been documented in prawn trawl bycatch in NW Australia (Hornborg et al. 2013). As of March 1st 2016, IUCN Redlist assessments were available for 64% (18) of the 28 species identified as bycatch in the Bahamian lobster trap fishery (IUCN 2016). The majority (15/18) were listed as "Least Concern", whereas two fish species (Mutton snapper, *L. analis*; Hogfish, *L. maximus*) were listed as "Vulnerable" and one (Nassau Grouper; *E. striatus*) as "Endangered". The two vulnerable species were a minor part of trap bycatch: *L. analis* comprised 1% of bycatch by weight at one location and *L. maximus* comprised 4% by number at another site. Both species were retained as bycatch and are of commercial and subsistence interests to fishers. Although
inclusion in the retained bycatch classification addresses the wastage aspect of bycatch, this type of bycatch still poses an ecological impact if of sufficient magnitude.

Of particular concern as lobster trap bycatch is the Nassau Grouper (*E. striatus*) due to its endangered status. Although *E. striatus* was not a common component of bycatch in the current study, it also occurs in other lobster fisheries. *E. striatus* was also identified as bycatch in the commercial wooden lobster trap fishery in Florida in 1993-1994 (Matthews et al. 2005). In that fishery, only three specimens were caught among 21,309 traps sampled but that extremely low occurrence was no doubt influenced by the low *E. striatus* population size in Florida. Still its appearance as bycatch in the commercial lobster fishery was considered a concern because the landing of *E. striatus* is prohibited in Florida (Matthews et al. 2005).

In The Bahamas a local assessment for Nassau grouper is available and there are management measures in place for the commercial fishery. The Nassau grouper population in The Bahamas is estimated to be fully exploited to overexploited and it is among the few existing populations of Nassau grouper within its historic geographic extent (Cheung et al. 2013). Management measures in place in The Bahamas include a minimum size limit of 3 lbs (1.36 Kg) and a closed season from December 1st to February 28th. Bycatch from the commercial fishery must comply with these measures. Nassau grouper landings as a component of trap bycatch in the lobster fishery are further curtailed due to the April 1-July 31 annual closed season for the lobster fishery, during which traps are to be removed from the water. The extent to which Nassau grouper that are caught as bycatch in traps and retained for consumption or released is unknown, and if released, the proportion that survives is unknown. Unlike crustaceans, the presence of an air bladder in Nassau grouper makes them susceptible to barotrauma. However, the lobster fishery takes place in shallow water (Gittens 2005) which greatly reduces the chances of barotrauma (Wilson and Burns 1996). Nevertheless, monitoring of the bycatch of Nassau grouper in lobster traps should be considered due to the endangered status of Nassau grouper and the temptation for fishers to retain it for local consumption.

Yet, estimates of bycatch per se are less relevant than estimates that incorporate the survival outcome of bycatch, which includes the rate of retention, dead discards, or discard survival (Vázquez-Rowe et al. 2012). My data from traps in the Bahamian lobster fishery revealed that dead discards of other taxa were 0.5-1.7% of lobster catches and comprised 1-5%
of bycatch (Table 6). A large proportion of bycatch was not wasted, because 31%-71% was retained for commercial or subsistence use, the ecological impact of this loss notwithstanding.

Although the estimates of bycatch presented in the current study for the trap fishery are informative, they may not be representative of the fishery in all seasons because observations took place during the final month of the open season when lobster catches are typically low (Dept. of Marine Resources, Govt. of The Bahamas records). As bycatch is typically calculated, and judged in relation to the weight of the target species, these rates and judgments are susceptible to fluctuations in the abundance of the target species and the bycatch species in the wild. When the abundance of the target species is low, bycatch is proportionally higher even if there is no change in the amount of bycatch. Likewise, when the abundance of the target species is high, bycatch appears to be lower. Thus, even when there are no changes in the weight of bycatch species, its relative proportion fluctuates intra-annually and inter-annually due to variation in lobster catch. In the current case, my estimates of bycatch are probably overestimates due to the waning of lobster catch late in the season, and thus should be considered a worst-case scenario for the fishery. Like with target species, it must also be borne in mind that unintended and undocumented mortality exists for non-target species. An example of this seen in crustaceans includes Nephrops fisheries in the Clyde Sea Scotland where injury of non-targeted crustaceans resulted in 30% lower survival overall (Bergmann and Moore 2001).

Because my estimates of trap bycatch are probably worst case scenarios, the IUCN lists most of the bycatch species as "Least Concern", and very few individuals were caught that are listed as "Vulnerable" or "Endangered" species by the IUCN, it appears that bycatch from the trap fishery does not appear to be of major ecological concern. In terms of magnitude, bycatch from the trap fishery does not appear excessive compared to international norms. However, a peculiar situation exists with respect to the consideration of trap bycatch from the trap fishery due to the presence of the red lionfish (P. volitans), an invasive species.

Lionfish, which are native to the Indo-Pacific, have spread throughout the Caribbean and south-western Atlantic and became established in The Bahamas around 2005; the species was first sighted in 1985 in Florida where it is thought to have been first introduced (Betancur-R et al. 2011). Negative impacts of the lionfish in The Bahamas continue to be discovered and include major reductions in the abundance and species richness of small fish at patch reefs (Albins 2013). Although Nassau grouper do not appear to eat lionfish and vice versa, the two species
interact by competing for space on reefs (Raymond et al. 2015). Via their consumption of a variety of small fishes and crustaceans (Morris Jr and Akins 2009), lionfish also reduce recruitment of reef fish in The Bahamas by 79% (Albins and Hixon 2008). The impacts of lionfish are exacerbated by the observation that lionfish attain larger sizes than in their native range (Whitfield et al. 2007) and that they also attain higher densities in invaded locations (Green and Côté 2009). As densities in some locations in The Bahamas can be eighteen times higher than those in other areas of the Caribbean (Green and Côté 2009), it appears that The Bahamas may be disproportionately impacted by the lionfish invasion. Like a number of other countries, The Bahamas has been attempting to spur a market for lionfish as a food source with the ultimate purpose of also increasing their removals from the environment via fishing (Moore 2012). The removal of lionfish as a result of the trap fishery thus contributes towards lionfish reduction efforts. Further, the link between bycatch and invasive species is not an entirely new perspective. It has been previously proposed that removal of terrestrial invasive species such as rats can be used to offset biodiversity losses due to bycatch of seabirds and turtles and that this would be consistent with the broader ecosystem tenets of the Convention on Biological Diversity (Donlan and Wilcox 2008).

I found that lionfish formed a major component of the bycatch associated with lobster traps: they comprised 53% of the bycatch by weight at SWTO and 11% at South Andros. Because lionfish are among the bycatch species that are retained by fishers, the presence of lionfish as bycatch in the trap fishery may help to ameliorate their negative ecological impacts. It would be useful to weigh the negative impacts of the bycatch of taxa other than P. volitans on the broader ecosystem with the positive impacts resulting from lionfish removal. Due to the major negative repercussions of the presence of lionfish in Bahamian waters, the benefits of using traps that capture lionfish might outweigh the negative aspects of trap bycatch. Lionfish also occur in casitas and could be captured by fishers of that gear, but the extent that they do so was not considered in this study.

Beyond the issue of bycatch, the size-selectivity of target species by competing types of fishing gear also has stock assessment implications. Among a variety of methods used for crustacean fisheries generally, and lobster fisheries specifically, size frequency based stock assessments date back to 1892 (Sparre and Venema 1989) and remain quite common because of the difficulty of aging lobsters (Smith and Botsford 1998, Punt 2003, Smith and Addison 2003,
Maunder and Punt 2013). When size selectivity data is utilized it often plays an important role in the eventual estimation of key exploitation parameters including the projection of maximum sustainable yield (Crone and Valero 2014). This includes integrated assessments where size frequency data is often integrated with multiple data types and its importance weighted (Punt 2015). Life history characteristics also influence the quality and type of data that can be used for stock assessments (Ono et al. 2015). For example, variability in recruitment is often reflected in cohorts represented by size frequency distributions (Smith and Botsford 1998). As such, assumptions about selectivity biases have far reaching consequences ultimately affecting the accuracy of stock assessments and the sustainability of the stock (Crone and Valero 2014).

The size frequency data typically utilized in lobster stock assessments in The Bahamas is fishery-dependent and primarily includes size frequency data of lobsters found at processing plants. Therefore, only legal sized lobsters would normally be found at processing plants because undersized lobsters are usually not accepted by processors. As such, the data collected at processing facilities does not include the sublegal size categories of lobsters that are present in casitas and traps, which has implications for the potential impacts on the survival, growth, and health of size classes that contribute to future fishery stocks. Given the importance of accurate assessments of the biases inherent in length-frequency data, it is necessary to assess the differences between the gear types used in the Bahamas lobster fishery. Unfortunately, logistical constraints and the low availability of lobsters in natural dens in the vicinity where I sampled casitas and traps precluded a direct comparison of the size-frequencies of lobsters found in natural areas. As a result, there is dependence on sampling gear in a similar manner to some deep water species that preclude SCUBA surveys (Goñi et al. 2003). Indeed, anecdotal observations by several long-term fishers on Abaco suggest that lobsters used to be much more prevalent in natural dens, mostly solution holes and rocky crevices, prior to the proliferation of artificial dens (casitas and traps) in the area. Although lobsters are not known to burrow or maintain solution hole dens, as do other species (e.g., *Epinephelus morio* (Valenciennes, 1828) - red grouper (Coleman et al. 2010)), their movements in and out of dens tend to keep den entrances free of sand. The presence of solution holes adds to habitat complexity. Habitat complexity enhances abundance and species richness (Diaz et al. 2003, Coleman et al. 2010). Now it appears that lobsters primarily use artificial dens in heavily fished areas where casitas and traps are
Numerous, and where natural dens may have filled with sand. This may be at the detriment of other species.

Despite my inability to estimate the size frequency distribution of lobsters in adjacent natural dens, I found no difference in the mean size of legal-sized lobsters (those ≥ 83 mm CL) found at casitas and retained in traps. Therefore, in the location studied, the size of legal-sized lobsters obtained from both types of fishing gear (casitas, traps) are similar and thus either equally representative or equally biased with respect to lobsters available in the environment. However, when the full size ranges of lobsters are considered (both legal and sub-legal sizes), the size frequency data from the two gear types differ, as may their usefulness for stock assessment. Casitas harbored a greater range and proportion of sub-legal size lobsters than did traps, which contained sublegal lobsters that were closer to the minimum size limit creating a size frequency distribution truncated at smaller sizes (Figure 13). Traps are known to be size selective and can vary in their catchability of lobsters by size and sex (Sparre and Venema 1989), therefore the presence of lobsters in traps is probably not as sensitive to fluctuations in the abundance of various size classes as are casitas. However, the size frequency distribution of the catch of lobsters from casitas may too be biased if predatory mortality on small juvenile lobsters is higher near casitas than in natural habitats, as has been observed at some locations (Gutzler et al. 2015). Nonetheless, the predictive use of sub-recruits in traps may prove to be useful as seen in Western Australia where, as far back as the 1980s, length-frequency data of sub-recruit *Panulirus cygnus* in traps has been invaluable as a predictor of catches approximately 1 year in advance (Caputi et al. 1995).

Promising results have also been obtained in Cuba where strong correlations were detected between catches, larval recruitment, and juvenile indices of abundance (Cruz et al. 1995). Yet, there have also been many failures in predicting fishery catch or adult abundance from estimates of larval or juvenile recruitment in many marine species (Sissenwine 1984), including lobsters because of post-settlement demographic bottlenecks (Wahle and Steneck 1991, Butler and Herrnkind 1992a, 1997, Briones et al. 1994) due to predation (Smith and Herrkind 1992) and the availability of suitable habitat (Afonso and Gruber 2007).

In conclusion, length-frequency data and its use in stock assessments are considered a key component of sustainable fisheries management. Consideration of this aspect of gear selectivity by managers as well as bycatch of undersized individuals and bycatch of other taxa is
important. Although these are just a subset of a myriad of factors that influence sustainable management, a lack of information on bycatch can lead to lost management opportunities and undermine the best of management efforts. Through the current study, it is now known that the use of hooks to extract lobsters from casitas does not negatively impact the lobster fishery via increased mortality of discarded bycatch of sublegal lobsters, though there is reaffirmation that handling mortality is real. It is also now known that multiple species of non-targeted taxa are present as bycatch in the lobster trap fishery that the magnitude of this bycatch is similar to levels reported in other fisheries globally, and that species considered "vulnerable" or "endangered" by the IUCN represent a small fraction of lobster trap bycatch. In an unusual twist, it appears that the bycatch of invasive lionfish in lobster traps is high and it is possible that levels of this bycatch could be large enough to benefit the marine ecosystem. There also appear to be opportunities to enhance stock assessment efforts by including sub-legal sized lobsters found in casitas and traps in stock assessment efforts.
CHAPTER IV
THE FEASIBILITY OF USING REMOTE SENSING TO DETECT CASITAS UTILIZED IN THE CARIBBEAN SPINY LOBSTER (PANULIRUS ARGUS) FISHERY IN THE BAHAMAS

INTRODUCTION
There are many examples of overfishing throughout the world (Cochrane 2000, Srinivasan et al. 2010). Many marine species (e.g., sea turtles, bluefin tuna, Nassau grouper) have been so heavily fished that they are now considered “critically endangered”, "endangered" or "threatened" (IUCN 2016), whereas others such as the Caribbean Monk Seal and Stellar Sea Cow were driven to extinction by over-harvest as long ago as the 1800s (Jackson 1997). Over the past several decades sustainably fished stocks have decreased globally from approximately 90% in 1974 to 69% in 2013 (FAO 2016). Overfishing threatens food security (Srinivasan et al. 2010) but is accelerated by the immediate need to provide protein for over 7 billion people. Due to the very real impacts of overfishing on the environment and human population, efforts abound to thwart over-fishing including major international agreements such as the 1995 FAO Code of Conduct for Responsible Fishing. Despite these efforts, some have opined that the limit to exploitation has been reached (Worm and Branch 2012) and that all currently fished taxa will be overfished by 2048 (Worm et al. 2006), although there is evidence of some improvement to a handful of major fish stocks since that dire prediction (FAO 2016). Nevertheless, it is clear that sustainable fishing is needed and equally clear that it is not easily achievable.

More data are required to assess the sustainability of many fisheries and there is great variability in the quality of stock assessments (Walters 1998, Caddy 1999), which is partly influenced by the quality and spatial availability of data (Worm and Branch 2012). Cost is also a factor. Simply stated, money and other resources are required to monitor, control, and accurately assess fisheries (Walters and Maguire 1996). This has resulted in disparities in the quality of stock assessments between developed countries and developing countries and between large-scale and small-scale fisheries (Mahon 1997, Costello et al. 2012). There is also evidence that smaller nations are more affected when overfishing occurs because their low per capita incomes
are exacerbated (Srinivasan et al. 2010). Yet another issue is the wise use of limited resources as it is not unheard of for developing countries to spend disproportionately more to conduct stock assessments when other aspects of management were perhaps a higher priority (Mahon 1997).

Beyond costs, other factors also influence the effectiveness of fisheries management. Ecosystem-based approaches, for example, highlight the need to look beyond individual species that are targeted by fisheries. In addition to ecosystem effects, other issues include economic considerations such as maximum economic yield, biological considerations such as levels of spawning stock biomass of target and non-target target species, and the effects of bycatch. Understanding the requirements of fisheries management can only diminish the likelihood of overfishing, but many challenges remain. One of the key shortcomings is the selection of inappropriate reference points brought on by shifting baselines (Pauly 1995); which can also be described as an absence of baseline information. After all, how does one accurately ascertain what percentage of spawning stock biomass remains without knowing what the initial spawning stock biomass or carrying capacity was? This is complicated by disagreement and difficulties in identifying what constitutes an appropriate management stock or management unit (Begg et al. 1999a).

Another major consideration is the role fishing gear plays in overfishing and the effect of that gear on the environment. For example, traps are can result in physical damage to the environment, may be associated with bycatch, and are subject to the insidious effects of ghost fishing when lost (Smolowitz et al. 1978, Chiappone et al. 2002, Matsuoka et al. 2005, Uhrin et al. 2014, Butler and Matthews 2015). However, the negative effects of trap use can be mollified to some extent with changes in construction materials, mesh sizes or slat spacing, and the number of traps and locations where they can be deployed. Although a lot is known about the impacts of traps, nets, trawls and a variety of other fishing gear, much remains to be done to mitigate their negative impacts. However, the limitation and avoidance of these impacts heavily depend on knowing what the impacts are, as well as where and how much fishing gear is deployed. Such is the case in the fishery for the Caribbean spiny lobster (P. argus) in The Bahamas, in which fishing gear referred to as "casitas" have exploded in use without regulation and with virtually no information on their abundance, distribution, or ecological effects.

Casitas (i.e., condos or pesqueros) are a type of fishing gear that is widely used in the fishery for P. argus in The Bahamas (and elsewhere in the Caribbean), representing a shift from
a predominantly trap-based fishery in just a few decades starting from the 1980s. However, the number of casitas in use is unknown (Gittens 2008) and unregulated. In The Bahamas, casitas vary in size and construction materials but are typically rectangular in shape with sides 1.8m x 1.2m (Figure 2, Chapter II). They also tend to have a top made of corrugated metal and 5cm x 15cm legs running along the 1.8m sides, thus resembling a table with very short legs that sits on the seafloor. Both ends of the casita are open; thus lobsters are not restrained from leaving, but tend to aggregate during the day under these artificial shelters. Casitas are fished by snorkelers or divers using hooka, who harvest lobsters from beneath casitas using long-handled hooks.

In some countries, including Cuba and Mexico, casitas have been utilized for decades and there is a strong and growing interest in their use throughout the wider Caribbean (Ramos-Aguilar 2003, FAO/WECAFC 2007). However, they are currently banned in other countries, such as the United States. The build-up in the use of casitas in Caribbean fisheries has been accompanied by an awareness that their use may have negative repercussions including recruitment overfishing (FAO/WECAFC 2007) or increased mortality of juvenile lobsters (Gutzler et al. 2015). Nonetheless, the appeal of this fishing gear is motivated by an increase in fishing efficiency that fishers may experience (CRFM 2005, Seijo 2008). The highly social nature of *P. argus* (Herrnkind et al. 1975, Kanciruk 1980, Horner et al. 2006b) leads lobsters to aggregate under casitas in unusually high densities, although this varies spatially and temporally (Eggleston et al. 1992). It is possible that casitas increase lobster production (FAO/WECAFC 2007) by increasing the amount of refuge available (Briones-Fourzán and Lozano-Álvarez 2001, CRFM 2005), thus enhancing the survival of juvenile size classes and perhaps their growth (Eggleston and Lipcius 1992). The possible benefits of using casitas have led to cautious optimism among countries that already use them and those that seek to enhance landings. This is reflected in the *St. George’s Declaration on Conservation, Management and Sustainable Use of the Caribbean Spiny Lobster (Panulirus argus)*, an agreement between the Caribbean territories (countries and British overseas territories) that comprise the Caribbean Regional Fisheries Mechanism (CRFM). Among many aspects of the management of *P. argus* fisheries, the Declaration calls for economic and ecological research to be conducted before casitas are introduced to a lobster fishery, and where they are already in use, that research should be conducted on their sustainability. In The Bahamas, casitas have increased in use since the 1980s and it is believed that they have exceeded the use of wooden lathe traps since the mid-1990s
Despite managers not knowing the ecological effects of this introduction and expansion.

Despite this possible shortcoming, The Bahamas has a number of management measures in place to protect the sustainability of the lobster fishery, but specific measures regulating the use of casitas are not included. Lobster regulations in The Bahamas include: a four month closed season (April 1-July 31), the banning of the possession of egg-bearing females, a minimum size limit (82.3 mm carapace length or 140 mm tail length), a “bag limit” for foreign-based recreational fishing vessels, and the prohibition of SCUBA for fishing. Though fishers using casitas also have to abide by these measures, unlike wooden lathe traps, there is no requirement to obtain permission from regulators to use casitas and there is no tracking of the number of casitas deployed. Casitas are also not removed from the sea when the fishery is closed, as there are no regulations governing this.

Regulators now seek to determine the ecological impacts of casitas and develop an approach to manage casitas in The Bahamas, but a number of factors make ascertaining their impacts daunting. These include uncertainties about the number and spatial distribution of casitas. Estimates of the number of casitas now in use in The Bahamas range from hundreds of thousands to over 1 million (Doerr 2014). These uncertainties are compounded by the expansive shallow water banks (> 116,000 km²) over which casitas might be distributed. The ability to accurately quantify and track the number and spatial distribution of casitas would be an aid to management. Doing so would allow monitoring efforts to be more targeted, would permit the quantification of how many casitas are lost each year, and would facilitate research on the basic ecological effects of casitas.

For example, casitas placed in seagrass habitats shade and destroy the seagrass beneath them and create sand "halos" around each casita due to the scouring and foraging activities of casita inhabitants (Nizinski 2007). When placed in lobster nursery habitats, casitas also increase rates of predation on juvenile lobsters that shelter in unnatural aggregations under casitas (Gutzler et al. 2015). In addition to ecological impacts, there may be economic considerations at stake with the widespread deployment of casitas. For example, maximum economic yield is often considered as a part of stock assessments (Grafton et al. 2007, Squires and Vestergaard 2015) from the perspective of fishing costs for individual fishers and on country-wide or stock-wide scales. Further, the Bahamian government has previously provided financial assistance to
fishers in the aftermath of hurricanes in instances when fishing activities were severely affected. An accurate and verifiable valuation of losses is a critical aspect of this. It is in a fisher’s best interest to document the number of casitas that they utilize as government assistance may be hampered if the number of casitas each fisher has is unknown. Thus, for multiple reasons the accurate estimation of the scale and spatial distribution of fishing gear such as casitas is crucial to sound fishery management and remote sensing is one technique for doing so.

Remote sensing has multiple applications that are relevant to environmental and fisheries management. As an example, the mapping of seagrass and coral reef locations, measurement of seagrass standing crop or biomass, measurement of sea surface temperatures, detection of ocean circulation and the management of marine protected areas are all relevant to the management of *P. argus* refuges, foraging grounds, and larval dispersal (Jupp et al. 1985, Mumby et al. 1997b, a, Dierssen et al. 2003, Lucieer and Pederson 2008, Hill et al. 2014). Indeed, some have opined that remote sensing is necessary for the efficient monitoring of coastal environments (Wang and Wang 2010).

Remote sensing relies on the collection of data about objects, area, and location from afar (Lillesand et al. 2014). It allows the repeated collection of data from a given location (Green et al. 1996) and offers an advantage for monitoring remote or hard to reach areas (Campbell 1996). Often the cost of traditional survey methods can be prohibitive (Mumby et al. 1999), even for a single visit to a remote location, so the ability to glean information from afar makes the expense of remote sensing more attractive especially as technology and computing power have improved and have become more affordable (Mumby et al. 2004).

Of particular relevance to the current study is the ability to detect objects in the marine environment. Modern day satellite-based remote sensing allows a variety of man-made objects as small as 1 meter in length to be detected (Inglada 2007, Cheng and Han 2016). This should be an adequate spatial resolution to detect typical Bahamian casitas that are approximately 1.8m x 1.2m in dimension, but detection in seawater poses challenges in comparison to measurement in the terrestrial environment (Chapron et al. 2008). Among these challenges is the need for satellite sensors with adequate radiometric and spatial resolution, as well as the ability to discern spectral characteristics unique to objects of interest. Doing so must also be achieved amid variation in the spectral characteristics of the background environment (e.g., bright sand versus darker seagrass) as well as variations in the angle of the sun, water depth, dissolved and particulate components of
seawater, and the roughness of the sea surface (Monahan and O'Muircheartaigh 1986)-not to mention variation in the spectral sensitivity of sensors (Kirk 1994, Campbell 1996). Nonetheless, in the clear (Lyzenga 1981), shallow waters of the banks of The Bahamas where casitas are typically deployed, advances in remote sensing technology, supported by advances in modeling, suggest that remote detection of casitas may be possible. Indeed, previous studies have already successfully used remote sensing to detect aspects of Bahamian bathymetry including the extent and variations in density of seagrass meadows (Louchard et al. 2002, Dierssen et al. 2003). Therefore, this chapter focuses on the potential development of satellite-based, passive remote sensing methodologies to estimate the number and spatial distribution of casitas while taking into account the vast and often remote nature of the casita-based lobster fishery in The Bahamas. I hypothesize that:

a) casitas can be detected in various benthic environments where casitas are typically deployed in The Bahamas, and

b) casita distribution and abundance can be detected using remote sensing technology after proper calibration and ground-truthing.

These hypotheses were tested using a modeling approach that included data collected from casitas at commercial fishing grounds in Abaco, Bahamas and the analyses of a sample remote sensing image of a section of Sandy Point, Abaco provided by Digital Globe.

METHODS

During October 2014, the characteristics of the visible portion of the electromagnetic spectrum in the water column above casitas and their background environment were obtained in Sandy Point and Marsh Harbor, Abaco (Figure 1; Chapter II). This was undertaken to estimate reflectance and contrast characteristics of casitas in commercial fishing surroundings. A diver-operated benthic bio-optical spectrometer (DOBBS) outfitted with three plane irradiance sensors was used to measure and record the spectral characteristics of casitas and their surroundings (Figure 14). The visible portion of the electromagnetic spectrum was modeled because visible light has the greatest penetration through the water column (Campbell 1996).

Data collected included spectral upwelling plane irradiance (E_u) and spectral downwelling plane irradiance (E_d). Three sets of readings were recorded over each casita to
calculate a representative mean for each casita. Similarly, background environment readings were recorded within 0.5 m around each of the four sides of each casita. Readings were collected from thirteen casitas and their immediate surroundings. These included six experimental casitas that I deployed and were constructed to the specifications of commercial casitas and seven casitas built and deployed by commercial fishers. There were seven and six sets of readings taken from casitas at Sandy Point and Marsh Harbour, respectively. The experimental casitas varied in how long they were in place on the seafloor ranging from 12-15 months. The length of time commercial casitas were in place is unknown but none appeared to be new.

Figure 14: SCUBA diver operating DOBBS during the collection of irradiance readings from the surface of a casita in Abaco, Bahamas. The corrugated surface of the casita and two cement blocks (weights) are evident. The background environment is a mixture of sparse *Thalassia testudinum* and sand.

Each set of readings was comprised of $E_d$ and $E_d$ measurements at 1nm wavelength increments in the visible light spectrum (390nm to 700 nm). These data were used to calculate reflectance ($R$) using the formula $R = E_d/E_d$ across the spectrum at each 1nm wavelength. The
characteristics of the background environment (e.g., light *Thalassia* seagrass, dense *Thalassia*, sand or shell hash) were also noted for each set of background environment readings. Water depth and time of day were also recorded for each group of casita and background readings.

These data were used to calculate diffuse attenuation coefficients ($K_d$) at each wavelength increment using Beer’s formula:

\[
K_d = -\ln \left( \frac{E_{d,\text{wand}}}{E_{d,\text{ref}}} \right) / \text{(wand height/100)}
\]

where $E_{d,\text{wand}}$ is downwelling plane irradiance measured at the surface of a casita or the background environment and $E_{d,\text{ref}}$ is downwelling irradiance measured at a reference height (wand height) of 1m. The $K_d$ estimates were then used to calculate an overall mean $K_d$ per 1nm wavelength increment based on all casitas from which readings were successfully obtained. These calculations were similarly completed for each background environment. The estimation of $K_d$ also involved a calibration process that included a comparison of known $K_d$ values of pure water (Pope and Fry 1997) to that of the seawater estimates for Sandy Point and Marsh Harbour. All seawater readings that appeared (literally and figuratively) to be clearer than freshwater were excluded from calculations.

After a number of intermediary calculations that utilized Beer’s law, absolute reflectance difference between casitas and various background environments at depths ranging from 0m to 10m were estimated. This involved using the formula:

\[
R_{\text{abs}} = R_{\text{cas}} - R_{\text{back}}
\]

where $R_{\text{abs}}$ is absolute reflectance difference, $R_{\text{cas}}$ is casita reflectance, and $R_{\text{back}}$ is background reflectance at each wavelength (390-700 nm) and each depth (0 – 5 m and at 10m). Contrast was then estimated at each wavelength and depth scenario using the formula $C = R_{\text{abs}} / R_{\text{cas}}$, where $C$ is contrast. This was conducted to determine the likelihood that satellite based sensors would be able to detect casitas in the water depths at which they typically occur.
**Image Analyses**

GPS readings of experimental casitas and fisher casitas were collected at commercial fishing grounds in waters near Sandy Point, Abaco. A sample panchromatic raster Tiff image of a 5 km x 5 km marine location that overlapped in location with the GPS readings was provided to me by Digital Globe. The image utilized was acquired with the Worldview 1 satellite on September 27th, 2014. As funds were not available to task a remote sensing satellite to obtain an up-to-date image of the study area, every effort was made to select an image that was already acquired by Digital Globe that minimized the time between the date that the image was acquired, the dates that GPS readings were collected and the date field verification of image analysis results were conducted. Image analysis techniques were conducted using ArcMap 10.2 and ENVI 5.1. These analyses included exploration of the brightness characteristics of the image’s pixels and an unsupervised classification process.

The image brightness exploration involved using ArcMap to determine the pixel brightness attributes of the GPS locations at a portion of the casitas within the Sandy Point sample image location. This initial step included a portion of casitas that were already known to be present. These unique brightness characteristics were then used in a “select by attribute” search of the image to detect other pixels that had the same characteristics as the casita locations. This was conducted with the expectation that other casitas had the brightness attributes of the known casitas and that the casitas would be found based on these characteristics. The locations of the resulting pixels that were selected were then compared to the locations of additional known casitas that were not previously used to determine brightness attributes. The image of Sandy Point was analyzed without altering the range of brightness values (>1100) of the original image.

Image analyses were also conducted using ENVI 5.1 remote sensing image analysis software. The image was first inspected visually with an overlay of the GPS locations of known casitas to ascertain whether casitas can be detected by eye. Thereafter, various unsupervised classification routines were conducted using tools available in ENVI. This included “k means” and “iso-data” classifications with classification schemes up to twenty-one unique classes. This was followed by visual inspections of the classified images with and without overlays of known casita locations to confirm whether known casitas were detected and to note the locations of suspected casitas not previously visited in the field.
The GPS locations of the suspected casitas were noted and subsequently (June 2016) visited to attempt to verify the presence of casitas. The verification process involved visiting the GPS location of the suspected casita. An observer then peered into the water with the aid of a waterglass. If a casita was not present, the vessel made larger and larger concentric circles up to a maximum of 50ft. This was to compensate for possible georeferencing errors. If a casita was found, the actual GPS location was noted and water depth was noted. Photographs were also taken to document the visual characteristics of the casita.

Image analyses also included the inspection of Google Earth images by eye. Google Earth imagery of Great Bahama Bank was inspected for objects that were the same shape and size as casitas.

RESULTS

The relative reflectance of casitas varied between 400nm and 700nm with relatively little variability between casitas based on confidence intervals (Figure 15). The mean relative reflectance estimates of three background environments (i.e., sand, dense Thalassia, and moderately dense Thalassia – a mixture of Thalassia and sand) varied in a similar manner. In other words, there was an increase between wavelengths of 400nm and 550nm, a plateau around 575nm, a distinct decline at approximately 650nm followed by a sharp rise to a maximum at 700nm. Compared to casitas, there was greater variability in relative reflectance at Thalassia, sand, and mixtures of Thalassia and sand. Nonetheless, there were negligible to no overlap of 95% confidence intervals for casitas, sand, and dense Thalassia. However, there was considerable overlap in reflectance confidence intervals of casitas and moderate seagrass (Figure 15). Most casitas were in a mixture Thalassia and sand. This resulted in 81% (42/52) of irradiance readings collected near casitas being a mixture of Thalassia and sand. Dense Thalassia and pure sand accounted for three (6%) and four (8%) of irradiance readings, respectively.

The relative reflectance of casitas and background environments also varied according to water depth and wavelength (Figure 16). Reflectance decreased as water depth increased with a tendency for reflectance to become indistinguishable at wavelengths >650nm in water ≥5m in depth (Figure 16). The mean absolute difference in reflectance between casitas and background environments also decreased as depth increased (Figure 17). Here the mean refers to the spectral
range considered (400-700 nm). Although casitas have lower reflectance than sand and a higher reflectance than *Thalassia*, the absolute difference in magnitude of the difference in reflectance was virtually identical between casitas and sand as well as casitas and seagrass. However, the difference between casitas and background environments comprised of moderate densities of seagrass was less distinct.

Figure 15: Relative reflectance of casitas, sand, dense *Thalassia* and a mixture of *Thalassia* and sand across the visible spectrum of wavelengths. Solid lines represent means while dotted lines represent 95% confidence intervals.

A close look at the variation in the difference in reflectance between casitas and *Thalassia* further revealed that the difference in reflectance was greatest (>0.08) between approximately 450-600nm wavelength (Figure 18). Whereas the difference in reflectance
between casitas and a mixture of *Thalassia* and sand was relatively low (≤ 0.04), but highest at wavelengths above 550nm (Figure 19). In sand, reflectance difference was negative: sand had higher reflectance than casitas (Figure 20). The magnitude of the difference varied by depth and was higher (≥ 0.05) in deeper waters between wavelengths that range from 400-550 nm.

Figure 16: Estimated variation in relative reflectance between depths and across the visible spectrum of light (400-750 nm) at casitas, *Thalassia*, a mixture of *Thalassia* + sand, and sand at Sandy Point and Marsh Harbour, Abaco.
Figure 17: Mean absolute reflectance difference between casitas and sand, casitas and a mixture of sand + *Thalassia*, and also casitas and *Thalassia*. The reflectance values used to calculate the mean were from the visible spectrum. Note that reflectance values were estimated for water depths of 0m, 1m, 2m, 3m, 4m, 5m and 10m. All other values depicted were interpolated.

Although a difference in reflectance existed between casitas and the background environments studied, contrast did not vary by depth within the depth range (0-10m) that I modeled. However, contrast varied according to wavelength (Figures 21-23).
Figure 18: Variation in the difference in reflectance between casitas and *Thalassia* with water depth

Figure 19: Variation in the difference in reflectance between casitas and a mixture of *Thalassia* and sand according to water depth
Figure 20: Variation in the difference in reflectance between casitas and sand according to water depth

As with reflectance and reflectance difference, relative contrast was higher in *Thalassia* and in sand but lower in a mixture of *Thalassia* and sand (Figures 21-23). Amid these general differences, contrast was higher toward the lower end of the visible spectrum. In each background environment, the magnitude of the difference in contrast was highest between 400-500nm wavelength and around 670nm.
Figure 21: Relative contrast of casitas at different light wavelengths in an environment that is predominantly *Thalassia*

Figure 22: Relative contrast of casitas at different light wavelengths in a mixture of *Thalassia* and sand
Figure 23: Relative contrast of light at different wavelengths between casitas and sand

**Image Analyses**

Exploration of a sample panchromatic image of a 5 km x 5 km area within commercial fishing grounds near Sandy Point, Abaco where casitas are known to occur showed that brightness values ranged from 0-1100 in Arcmap. Brightness values at the location of known casitas was narrow and ranged from 198-202. A “select by attribute” search for other pixels in the image detected many other pixels throughout the image that were within the same 198-202 range. This resulted in the selection of 9,411,593 pixels (8.4%) of the total (111,393,180) pixels. Based on an examination of the metadata associated with the image, pixels were 0.0000045 x 0.0000045 degrees in size, which is the equivalent of 0.5m x 0.5m in Sandy Point. This was adequate spatial resolution to detect 1.8m x 1.2m casitas. In addition, casitas should be
approximately 12 pixels in size (4 pixels x 3 pixels). As such, if all of the 9.4 million pixels that were within the 198-202 brightness range were truly from casitas, then there should be approximately 784,299 [calculation: 9,411,593 ÷ 12] casitas within the image. The selected pixels were at times isolated (surrounded by non-selected pixels), in small clusters and large clusters of pixels.

A visual inspection of images classified in Envi 5.2 using K-means and Iso-data routines did not result in pixels at known casita locations appearing unique in terms of color class or pattern of colors. However, a visual inspection of the non-classified panchromatic image showed a handful of locations that were consistent with the size, shape and brightness of casitas inside a darker Thalassia background (Figure 24). Three of the five suspected casitas also appeared unique in the aforementioned classified image.

Furthermore, the colors of the suspected casitas were not unique in the classified image, however, the pattern of colors was at two of the suspected locations. At two of the casitas there was a blue center with a green ring/halo around, all of which were surrounded by red (Figure 25). The red pixels were believed to be dense Thalassia. The suspected casitas were equally distinct by eye in the panchromatic image whereas only two were distinct in the classified image.
Figure 24: Panchromatic image of an area in Sandy Point, Abaco with suspected casitas. The single bright spot in the center of each red square shows the suspected locations. The inset shows the larger 25km² from which the area delimited by a green rectangle was obtained.
Figure 25: Classified version of Figure 24. Colors resulted from an ENVI 5.2 Isodata unsupervised classification tool applied to the original panchromatic image. A total of five colors resulted with each color representing unique brightness characteristics. Black rectangles indicate the location of suspected casitas at their center. Suspected casitas 1 and 2 are the most distinct with a blue center, a green halo and then a large red area that is suspected of being dense *Thalassia*. Blue is likely to be bare sand and light green is likely to be a mixture of *Thalassia* and sand.

The sites of suspected casitas seen in Figures 24 and 25 were visited for verification of the presence of casitas approximately 22 months after the date of the satellite image. Casitas 2
and 4 were present. As suspected Casita 5 was within 20m of Casita 4, it is also possible that the casita found was 5 instead of 4. At casita 1 there was an ovoid shaped sandy area in the midst of dense *Thalassia* that may have been where a casita was located. Casita 2 was made of an atypical metal material with corrugations sufficiently far apart that the casita was predominantly flat (Figure 26). It was also estimated to be smaller in size than a typical 1.8m x 1.2m casita and was surrounded by dense *Thalassia* in waters 1.5-2.0m deep.

![Figure 26: Casita 2 detected by remote sensing in Sandy Point, Abaco. The casita was made of an atypical metal surface and had algae growing on much of it. It was also surrounded by dense *Thalassia*.](image)

There was a clearing of sand surrounded by moderately dense *Thalassia* at the site of suspected casita 3. Casita 4 (possibly 5) was found in shallow waters approximately 1.5-2m deep.
and was comprised of a flat galvanized metal material. It was found inverted at the time it was visited and was surrounded by moderate *Thalassia* (Figure 27).

![Figure 27: Casita 4 detected using remote sensing in Sandy Point, Abaco. The casita was found inverted and in waters 1.5-2m deep. It was also surrounded by moderately dense seagrass. A cement block is also seen and is typically used as a weight.](image)

Inspection of a Google Earth Image image by eye yielded a number of suspected casitas. The suspected casitas were bright compared to the background marine environment and were rectangular in shape (Figure 28).
DISCUSSION

Casita use in the Bahamian *P. argus* fishery has gone largely unregulated for decades. Although this does not mean that the fishery is unsustainable, it makes verifying sustainability more complicated. The fact that casita use is so common in The Bahamas and fishing grounds so vast (Gittens 2008, Doerr 2014), adds to this challenge. The deeply rooted and pervasive use of casitas is also likely to make it difficult to implement changes in how this gear is managed. The periodic abundance of lobsters found within casitas has convinced many fishers that casitas increase lobster production and, given that they are increasingly (and rightfully) given an opportunity to offer opinions on how fisheries should be managed, may resist restrictions on the use of casitas (Berkes 2009). There is a need to evaluate the ecological and economic effects of casitas as a part of an overall effort to assess sustainability. The possibility that remote sensing might allow the number and spatial distribution of casitas to be determined is appealing for fishery managers in The Bahamas due to the vast and often remote nature of the fishery and the tradeoffs in management costs for remote sensing versus traditional fishery monitoring strategies. The results of the current study suggest that casitas used in The Bahamas are detectable in the marine environment under certain circumstances, and thus provide new options for the monitoring and management of this type of fishing gear.
A number of factors facilitate the detection of casitas in The Bahamas. First, the lobster fishery in The Bahamas is primarily a shallow water fishery, much of it concentrated in the shallow water of the Great Bahama Bank and Little Bahama Bank (CFMC and CFRAMP 1999, CRFM 2005). Because upward flux of light decreases with water depth (Duntley 1963), the fact that commercial fishing in The Bahamas is largely in shallow water, sometimes as shallow as 1.5 m, facilitates detection of casitas. The waters of The Bahamas are also clear (Lyzenga 1981), thus facilitating the transmission of light and further facilitating remote sensing.

Commercial fishers in The Bahamas also typically set casitas in seagrass or a mixture of seagrass and sand. Without considering the complications brought on by water depth, the relative reflectance of casitas was distinctly brighter than dense seagrass throughout the visible spectral range (Figure 15). The reflectance measurements obtained in seagrass (Thalassia testudinum) during the current study had a mean below 0.1 throughout the visible spectrum. A similar range of reflectance values (i.e., <0.1 throughout visible spectrum) were detected near Lee Stocking Island in a previous study (Louchard et al. 2002). However, Dierssen et al. (2003) found higher reflectance near Lee Stocking Island with R < 0.1 for parts of the visible spectral range but exceeding 0.1 (while remaining below R = 0.2) for some portions of the spectrum. In the Dierssen et al. (2003) study, reflectance also varied with seagrass density and was affected by the method of estimation. Nonetheless, reflectance in seagrass was low in the current study and the two other studies mentioned. Thus, it is likely that there is minimal variation in reflectance in seagrass by location in The Bahamas.

An unknown fraction of Bahamian casitas are deployed on sand bottom and, because they are constructed of light-weight materials, they are also susceptible to movement by currents and may shift into sandy locations. Nonetheless, even in an environment dominated by sand, casitas should also be detectable using remote sensing because they have a markedly different reflectance than sand; in this case, sand is brighter. The difference in absolute reflectance between casitas and seagrass-or sand-dominated environments also applies to a variety of casita types that differ in construction. I obtained reflectance readings from casitas made of several different materials, yet despite this diversity in construction the variability in reflectance among casitas was minimal as evidenced by relatively narrow 95% confidence intervals (Figure 15). There was also negligible to no overlap in the confidence intervals between reflectance measurements of casitas and seagrass, or casitas and sand. Results may differ in other countries
as all of the casitas sampled in the current study were primarily made of metal whereas concrete is used in other territories (Arce et al. 1997, Gutzler et al. 2015). The Louchard et al. (2002) study had similar results as various sand types had distinctly higher reflectance than seagrass and other backgrounds. This was similarly observed by Dierssen et al. (2003) who demonstrated the highest reflectance values (at times exceeding R = 0.5) for sand compared to R < 0.4 for the current study. As such, there is variation in reflectance of sand by location (and or possibly time) and type of sand, however sand consistently had higher reflectance than other backgrounds. As sand size and composition are known to vary with wave energy (Dean 1973), reflectance may also vary with wave energy. Wave energy also varies by location.

Most casitas in the current study were found in a mixture of moderate to sparse seagrass that intersperse with patches of sand. If reflectance of dense seagrass and of bare sand are considered the ends of the background reflectance continuum relevant to casitas, the resulting reflectance of a mixture of sand and seagrass would fall somewhere along that continuum. This is problematic because the reflectance of casitas and that of a mixture of sand and seagrass are less distinguishable, even without consideration of the effects of water depth on substrate reflectance. Louchard et al. (2002) also observed reflectance values that were intermediate between bare sand and dense seagrass in locations where sand was visible among seagrass.

Although the reflectance of casitas, sand, and seagrass varied in magnitude, the similarities in the pattern of variation in reflection at each wavelength were notable. For example, in each background environment there was an abrupt decline in reflectance followed by an increase at approximately 665-685nm. This demonstrates that, although the background environment influences spectral signatures, other aspects of the water column have a major influence on reflectance (Haltrin 1999). The detectable reflectance of the seabed and objects on the seabed are greatly affected by dissolved components in the water column (Lyzenga 1981). The decline in reflectance that I observed around 675 nm is likely due to absorption by chlorophyll (Moss and Rock 1991, Gitelson et al. 1996) present in the water column throughout all of my study areas. As dissolved components in the water column are likely to vary by location due to runoff and its associated nutrient inputs, reflectance may vary by location in the Bahamas due to variations in proximity to land and levels of rainfall within The Bahamas. These effects would be limited for much of the distal fishing grounds that are minimally affected by runoff from major landmasses. A complete absence of riverine runoff into the marine
environment in The Bahamas also limits the effects of runoff and favors the use of remote sensing in The Bahamas in contrast to other countries where *P. argus* and casitas are found. However, it is possible that the effects of dissolved components on reflectance is negligible compared to other factors such as water depth and reflectance difference in other countries. Location specific investigations that consider the various influences of remote sensing would be best.

The magnitude of the effects of water and its constituents on the transmission and absorption of light incident on the seafloor and emerging from the sea surface also vary according to depth. For example, reflectance declines with increasing water depth, as I observed when measuring reflectance from casitas and for each of the background environments. Reflectance is estimated to be at its maximum at 0m (sea surface) and gradually diminishes throughout the visible spectrum as depth increases (Figure 16). At each water depth modeled in the current study, the differences in reflectance between casitas and the different environments persisted as water depth increased. However, the difference between casitas and the background environment decreased as depth increased (Figures 17 - 20), which means that as water depth increases the ability of satellite-based sensors to discern casitas on the seafloor diminishes. The notable exception to this pattern concerns detection of casitas lying on a mixture of sand and *Thalassia*, which is equally challenging regardless of water depth. The difference in reflectance between casitas and a mixture of sand and seagrass is the same (*R*<sub>abs</sub> < 4) whether in water < 1m or in 3.5m of water (Figure 17).

As Figures 16 shows, relative reflectance declines as water depth increases throughout the spectral range modeled. It suggests that a sensor that utilizes the spectral range would be adequate for detecting casitas. This includes panchromatic imagery. However, it is also evident that the effects of changes in depth on light reflectance are not the same at each wavelength (Figure 16); reflectance tended to be highest at wavelengths between 400nm and 600nm regardless of substrate. A closer look at an even narrower spectral band shows that at the surface the maximum reflectance is ~ 590nm with a gradual shift in maximum reflectance at deeper depths, with a maximum at ~500nm in 10m deep waters. As such, sensors that target the 400-600 nm or 550nm – 600nm wavelength may achieve better results than panchromatic imagery. Many satellites are equipped with sensors that can target or are most efficient within these spectral ranges (Klemas 2009). Nonetheless, panchromatic imagery is advantageous as it typically has
better spatial resolution (Pohl and Van Genderen 1998), a major issue when attempting to detect 1.8m x 1.2 m casitas. In addition, not all satellites are available or equipped for marine applications. Satellites that acquire high resolution imagery between 0.5m-1.8m were common up to 2009, when there were approximately 18 civil satellites in use with this capability (Klemas 2009). However, poor spatial resolution can be overcome by higher contrast (Schowengerdt 2006), by utilizing different sensors for waters of different depths, or through the simultaneous use of multiple sensors. The use of multiple sensors is becoming more common (Chavez et al. 1991) and it allows multispectral lower resolution data to be integrated with high resolution panchromatic imagery with the outcome being an improvement in the resolution compared to images produced with just the multispectral sensor on its own (Pellemans et al. 1993, Pohl and Van Genderen 1998). There are also several methods for combining images from different platforms (Chavez et al. 1991), thus allowing analysts to select an option that best suits their needs.

Taking into account both background habitat and water depth, it is clear that the difference in reflectance between casitas and seagrass varies with depth and wavelength in a similar manner to absolute reflectance (Figure 18). This is a key consideration when discerning casitas in seagrass. The difference in reflectance between casitas and a dense seagrass background is highest between 400-600nm down to a depth of 5m. However, that relationship changes when casitas lie within the most challenging background environment (i.e., seagrass of moderate density interspersed with sand), where the difference in reflectance is greatest between 600nm and 700nm in waters ≤ 1m. These results suggest that employing more than one sensor to detect casitas is a better approach because spectral sensitivity varies among sensors. The use of multiple sensors with varying spectral and spatial resolution may also provide the most accurate and detailed images possible (Keys et al. 1990, Pohl and Van Genderen 1998).

The variations in estimated reflectance and reflectance difference between casitas and background habitats also change with water depth, so analysts must take into account water depth to ensure accurate identification of casitas. For example, a failure to detect casitas in waters ≤ 2m is more conclusive than a failure at 4m. As seen in Figure 17, a change in water depth from 0m to 2m results in a 50% change (from four to two) in reflectance difference between casitas situated within moderate seagrass. A similar change in reflectance difference occurs across the same depths when casitas are deployed in sand. Given the effect of water depth
on differences in reflectance, fluctuations in daily and seasonal tidal height might also be considered when comparing images from different times or when tasking satellites for new images.

Although many factors affect the quality of a remote sensing image, contrast plays a fundamental role in the eventual resolution (spatial and radiometric) of an image (Campbell 1996). Contrast is generally defined as “the difference in brightness between an object and its environment” (Campbell 1996), therefore the difference in reflectance between two objects is a measure of contrast and it varies with water depth and among backgrounds. Contrast can also be defined from in terms of the magnitude of reflectance difference in relation to overall reflectance. As seen in the Methods it was calculated in the current study as $C = R_{\text{abs}} / R_{\text{cas}}$. As seen in Figures 21-23, using this definition, contrast followed the same pattern as reflectance and reflectance difference to a large extent. It was greatest in seagrass and in sand but was lower in moderate seagrass. It also varied within the spectral range studied. Thus, the earlier conclusion, that multiple sensors (with differing spectral sensitivity) should be considered, still holds. Unlike reflectance, and reflectance difference, contrast (the ratio of reflectance difference to casita reflectance) did not vary according to depth, at least at the depths relevant to this study. Although this is a favorable outcome, it does not negate the challenges of detecting the difference in reflectance between casitas and the environment as water deepens.

Based on the distinct differences in reflectance between casitas and dense seagrass, and between casitas and sand, examination of the sample high-resolution panchromatic image should permit detection of casitas. A contrast stretch is commonly performed for such applications (Campbell 1996, Mather and Koch 1999, Schowengerdt 2012), however, this was not necessary because the image had a high radiometric contrast range; there were 1100 pixel brightness levels in the image. The pixel brightness characteristics at GPS locations of known casitas were also unique and narrow in range, suggesting that casitas should have been easily detected. However, based on the estimated number of pixels that should comprise a typical casita (12 pixels), the estimated maximum number of casitas (784,299) was too high. If true, then casitas would cover 8.4% of the seafloor in the Bahamas. Any field observation of the seafloor around Abaco where this study was conducted would certainly contradict this estimate; around Abaco where the satellite image was acquired, casitas are few and far between.
Many of the pixels suspected of being casitas were too small, whereas other pixels occurred in large clusters too big to be casitas and others appeared as irregular shapes. These results suggest that more advanced analyses based on pattern recognition are needed. Pattern recognition strategies are widely used in remote sensing and help to refine analysis projections (Melgani and Bruzzone 2004). For example, pattern recognition assisted in the detection of propeller scars within a seagrass environment in Florida (Oguslu et al. 2014). In the Oguslu et al. study, pixels from possible propeller scars were firstly detected and classified as being from a propeller using a sparse coding technique with the knowledge that false positive locations were also selected. In a second step, pixels suspected of being propeller scars were then further refined based on a shape and direction. This two-step process resulted in 90% accuracy in the detection of propeller scars within seagrass (Oguslu et al. 2014).

One caveat to the results presented in this study is that brightness levels attributed to casitas may be inaccurate as they depend on accurate horizontal geolocational accuracy. Though, in practice, this may be affected by many a number of factors including the possibility that GPS readings were not collected directly over casitas in the field, GPS readings were collected directly above casitas as far as practical in the current study. However, notably, descriptions of product offerings for Digital Globe state that horizontal geolocational accuracy is between 5m and 23m. Nevertheless, it is notable that a simple visual inspection of the image by eye also did not result in casitas being detected at or near the GPS readings, thus suggesting that georeferencing was not the main factor that prevented casita detection.

A visual inspection of the image for any objects that could be casitas yielded a handful of locations that were suspected to be casitas (Figure 24). Indeed, two casitas (2 and 4) were found to be present. Although the particular image analyzed was selected to minimize the time between the date that the image was acquired by a satellite (September 27th 2014) and the likely time that field verification of results could be conducted, more may have been detected if my field validation had taken place sooner. During the current study the length of time experimental casitas remained in place in Sandy Point, Abaco (i.e., without being moved by the author) varied from 3 months through the duration of the study (4 years). If this length of time is typical for casitas in Sandy Point to remain in one location, then the absence of casitas during verification does not confirm that casitas were not present at the time the image was acquired. Moreover, in
the cases of casitas 1 and 3, there were sand patches present that could have resulted from the death of seagrass resulting from casitas that were once present.

Yet, it cannot be ignored that many casitas that were present were not detected in the satellite image. This discrepancy may be due to several factors including: casita construction materials, background habitat, water depth, and horizontal geolocational accuracy, among others. In particular, as previously mentioned in this dissertation, most casitas were found in a mixture of seagrass and sand, an environment in which casitas are hardest to detect. Nonetheless, the fact that some of the casitas detected in the image were present supports the conclusion that remote sensing can be used to detect casitas. This is further supported by suspected casitas seen in Google Earth imagery (Figure 28). The length of time between the date of the Google Earth image (CY 2004) and the present study precluded verification but the rectangular shapes, proximity to each other and brightness strongly suggest that the casitas were present. All in all, the results and shortcomings also show that additional work is required if remote sensing is to gain practical and efficient application for the detection of casitas in The Bahamas. In particular, it is desirable for the detection of casitas to be as automated as possible. If there are indeed 200,000-over 1,000,000 casitas in use (Doerr 2014), then enumeration by eye using remote sensing images is not ideal.

Among the casitas that were found to be present, Casita 2 was located in dense seagrass, thus facilitating contrast between the casita and its background, hence detection of this casita. Casita 4 was also detected via remote sensing even though it was positioned in moderate-to-sparse seagrass. This demonstrates that casitas can be detected in a mixture of seagrass and sand in some instances. Notably, both casitas were in especially shallow water (< 2m), further enhancing the chances of their detection. The rectangular shape and size of Casitas 2 and 4 also played a role in their detection, supporting my earlier suggestion that pattern recognition could play a role in casita detection using remote sensing technologies. The color classification information provided by the ENVI 5.2 Isodata unsupervised classification tool applied to the original panchromatic image are a promising starting point, but also confirm that further investigation of the applicability of this analysis is needed.

The utility of remote sensing to The Bahamas may also go beyond the detection of casitas. For example, remote sensing has also been used to inventory natural resources (Wang et al. 2009) and my results confirm the usefulness of remote sensing for the detection of seagrass
over large seascapes. This was evident in the classification results and in picture confirmations. These kinds of data may also find application in the monitoring and management of the lobster fishery because seagrass is an important settlement and foraging habitat for *P. argus* (Herrnkind and Butler 1986, Childress and Herrnkind 1994, Cox et al. 1997, Lipcius et al. 1998). Thus, remote sensing might offer a way to track the availability or variability in lobster nursery areas and feeding grounds. Indeed seagrass presence has been successfully detected during previous remote sensing experimentation in The Bahamas that, similarly, combined in situ reflectance data and radiative transfer modelling (Dierssen et al. 2003).

The ability to monitor seagrass density and its geographic extent utilizing remote sensing may be particularly useful to The Bahamas where the population dynamics of *P. argus* are thought to be more locally constrained than in most areas of the Caribbean. *P. argus* is broadly distributed in the Caribbean with genetic linkages that span its entire range from Bermuda to South America suggesting that lobsters throughout the Caribbean are a single large stock that could be considered a single management unit (Silberman et al. 1994). However, more recent genetic analyses using microsatellite genotyping and direct (e.g. kinship analysis) and indirect genetic analysis techniques (e.g., kinship analysis and FST-based analyses of genetic differentiation, respectively) have revealed differences in genetic population structure among smaller regions (e.g., portions of Mexico, Bahamas, etc.) (Truelove et al. 2015). Those genetic results are in accord with biophysical models of larval dispersal, which predict that some sub-regions within the Caribbean are strongly linked through recruitment while others tend to retain larvae and are therefore largely self-recruiting (Kough et al. 2013). This appears to be the case for The Bahamas (Kough et al. 2013). Yet, whether the scale of management required is from Bermuda to Brazil or simply comprises the Great Bahama Bank and Little Bahama Bank, remote sensing makes management at these scales much more plausible. Remote sensing allows ecological assessments to take place at much larger scales than methods based on field visits allow (Kerr and Ostrovsky 2003).

Technological advances, cost reductions, and the variety of ecological and coastal management applications that remote sensing can be used for have made it a more viable option and a desirable management tool (Klemas 2009). The use of remote sensing in ecological studies has been increasing in use and has even been described as indispensable (Kerr and Ostrovsky 2003). Ultimately, the use of remote sensing to detect casitas in The Bahamas appears possible.
However, greater refinement is needed to make the detection of casitas using remote sensing more accurate and possibly even routine. The use of this technology might also bring additional monitoring and management options for *P. argus* fisheries that were not previously possible.
CHAPTER V
CONCLUSIONS

There are many factors that influence fishery sustainability, from broad considerations such as the role of fishing as a part of an overall sustainable development plan, to narrower influences such as the percentage of virgin spawning stock biomass of a target species that remains after a single fishing season. The multitude of factors that influence sustainability allow many indicators to be used to assist with determining whether fishery sustainability is likely. This dissertation focused on key biological effects of casitas on the sustainability of the P. argus fishery in The Bahamas.

One of the key biological effects investigated was predation. A study in Florida confirmed that predation was higher for juveniles at casitas in nursery areas (Gutzler et al. 2015). Another study in Mexico had mixed results for different size classes (Eggleston et al. 1992). These inconsistent results between locations confirm that fishery-specific research is needed. The current study supports this conclusion. Indeed, results were different between my study sites in The Bahamas. There was no evidence that casitas increased mortality of lobsters at casitas compared to natural areas. Casitas also did not increase the mortality of juvenile (< 80 mm CL) lobsters. The implications for management are that casitas can safely be used in juvenile and adult locations. Nonetheless, there is a chance that results may differ at other locations within The Bahamas. This is because risk of predation may vary between locations. Notably, the current study also showed that predation risk was higher at Marsh Harbour compared to Sandy Point, independent of casitas. Thus, despite differing overall predation risk between locations, casitas did not increase predation risk beyond naturally occurring levels.

Another issue to consider is the influence of casita design on predation. Casita design is relevant because it influences the size and types of predators that can access lobsters within the recesses of casitas. Therefore, there is a chance that the similarities in casita design between Sandy Point and Marsh Harbour and other locations in The Bahamas may result in similar effects (or absence thereof) on predation throughout The Bahamas.
This dissertation also compared an alternative prominent fishing gear (wooden lathe traps) that was once the dominant fishing gear and that would likely replace casitas if the use of casitas was abolished. Among the factors considered (i.e., growth rates, injury rates, blood protein levels and disease rates) it was evident that traps were more of a risk to the fishery. Lobster blood protein levels were significantly lower in traps and signs of shell disease were only seen in lobsters found in traps. Casitas did not appear to reduce lobster production via effects on growth rates, injuries or disease, whereas traps would be problematic if allowed to continue to fish during the closed season or if lost. Similar traps used in Florida fished for over 500 days once lost in some instances (Butler and Matthews 2015). Although the length of time a trap takes to degrade varies, the lobster fishery in The Bahamas would stand to benefit if biodegradable panels became mandatory for wooden lathe traps. Consideration would have to be given to how this would be implemented. For example, a single wooden panel might be the best compromise instead of the entire trap so that the cost of fishing does not become excessive. The length of time before a panel degrades would also have to be determined. Based on the current study, shell disease was detected in all lobsters after six weeks of confinement in a trap. Consideration would have to be given to making all traps have at least one panel that degrades by six weeks. Nonetheless, it would perhaps be more practical for biodegradable panels to last for nine months as this is the length of the fishing season. The traps would be fished all season long and would have lobsters removed regularly during the open season. As such, there may be no advantage to utilizing a shorter period of time for biodegradation. Further, most traps would cease fishing at the beginning of the closed season if a panel degrades after nine months. This is because new traps are typically deployed at the beginning of the nine-month open season.

Another aspect of the use of casitas considered was the use of the lobster hook. There was no evidence that survival of lobsters mistakenly hooked by fishers and then released differed from unhooked lobsters. Thus the rapid collection of lobsters from casitas using lobster hooks, an aspect of the efficiency of fishing with casitas in The Bahamas, does not appear to have a major impact on lobsters that are released. Nonetheless, this is highly dependent on fishers not inflicting a second and deadly blow to the ventral surface of the cephalotorax with the knife-like end of a lobster hook (Figure 9; Chapter II) before releasing lobsters. A comparison to traps is once again useful here. For example, it is known that there is cryptic / undetected mortality due to injuries associated with traps as seen in lobsters and other decapods (Coggins et al. 2007) and
a percentage of lobsters are likely to die from exposure (Matthews 2001) even when fishers return undersized or egg-bearing lobsters to the sea after hauling traps. This puts the use of lobsters hooks and casitas in a favorable light but is highly dependent on the ability of fishers to quickly discern whether a lobster that they initially hooked is undersized.

Yet another consideration is the size of lobsters found within each gear type. Though fishers depend on the ability of fishing gear to attract or retain lobsters, a failure to attract or retain sub-legal sized lobsters would favor sustainable management as this would reduce the interaction between fishers and lobsters. Casitas contained a larger size range of lobsters than traps, including a greater proportion of undersized lobsters. But all lobsters retained in traps would only be released after being removed from the trap and after exposure to air during hauling, whereas at casitas this is most likely to occur if lobsters are near to the legal size. Presumably, with casitas, fishers are able to easily determine that smaller juveniles are sub-legal in size without having to hook and release them. Further, sub-legal lobsters may be left in traps to attract other lobsters, thus increasing chances of shell disease and lowering blood protein levels among lobsters that enter the traps. Thus it can be argued that casitas have a smaller impact on sub-legal sized lobsters although they may attract a greater proportion of juveniles than traps. It thus appears that, based on susceptibility to disease and the other likely impacts on sub-legal sized lobsters, traps are a greater, but manageable, risk to sustainability.

Consideration of the effects of casitas and traps also goes beyond the target species. Bycatch is a major consideration (Hall et al. 2000, Davis 2002). Whereas bycatch of non-target taxa associated with casitas is suspected to be negligible, there was appreciable bycatch associated with traps. This favors the use of casitas compared to traps. This is further supported by the fact that lobsters are typically of higher commercial value compared to other fisheries resources. As a result, it is less likely that fishers will divert attention from targeting lobsters at casitas to target other species. In addition, it would be difficult to catch fish using a lobster look; however, broader impacts associated with casita use have to be considered. For example, the shading of seagrass by casitas or the leaching of metals into the marine environment from the galvanized metal from which casitas are made should be considered. Likewise, the effects of disabled traps or trap pieces were not considered as a part of the current study. These impacts are likely to be of significance if the effects are similar to the effects of marine debris on the physical environment (Chiappone et al. 2002) found in other lobster fisheries.
Yet another consideration, related to bycatch in the Bahamian lobster fishery, is the ecological effects of lionfish. The fact that traps also retain lionfish and traps must be hauled for fishing to take place, provide an opportunity for trap fishers to mitigate the effects of lionfish through their regular fishing activities. Fishers that use traps would have to extend themselves less to retain lionfish from traps than they would if they caught lionfish at casitas. Further investigation is needed, but, considering the negative ecological impacts of lionfish, the regular removal of lionfish from the sea using lobster traps may help to offset the negative ecological impacts of trap bycatch. There is also potential for fishers that utilize casitas to capture lionfish found nearby; however this would require a special effort as fishers typically utilize lobster hooks in conjunction with casitas. Lobster hooks are a less than ideal tool for capturing or killing fish. Nonetheless, it is common knowledge (among fishers and scientists that encounter lionfish) that lionfish are much easier to catch compared to other fish due to their tendency to not flee from divers. However, this relies on fishers taking the time to target lionfish at casitas whereas they are commonly retained in traps unless fishers avoid deploying traps in areas where lionfish occur.

The relative amounts of each gear type should also be weighed when comparing the impacts of casitas and traps and when evaluating their combined impacts on sustainability. During the 2015-2016 fishing season, there were approximately 47,000 traps for which licenses were obtained. However, the number of casitas in use is at minimum five times this at 200,000 and potentially over 1 million are in use (Doerr 2014). As such, the proportion of lobsters that interact with and are affected by traps is very likely to be small compared to those that interact with casitas. Even so, the current study did not detect any biological effects of casitas on lobsters. This makes it likely that there is a limited combined impact of casitas and traps. Nevertheless, beyond the aspects of bycatch considered in the current study, cryptic mortality and ghost fishing are known to exist in trap fisheries. These effects can be minimized with prudent management, including ensuring that fishers remove their traps from the sea during the closed season and the introduction of a requirement for wooden traps to have a biodegradable panel.

The number of casitas in use is also relevant because there may be effects outside the scope of the current study. It is not uncommon for new risks associated with gear types to be discovered, for example, the risk of shell disease associated with the use of traps in The Bahamas.
in the current study. The risks associated with casitas have just begun to be unraveled, thus not knowing the number of casitas and their spatial distribution remains a problem. Tools like remote sensing may be a viable option for monitoring the lobster fishery in The Bahamas. The results in this dissertation show that casitas can indeed be detected using remote sensing, though there was limited success. Casitas that were thought to be present were not detected using remote sensing. Unfortunately, a long time (22 months) elapsed between the taking of the satellite images and my ground-truthing of the presence of actual casitas, which complicates interpretation of these results. The casitas that were detected were also detected in the image by eye. The use of remote sensing to detect casitas would be more practical if detection was automated. Counting hundreds of thousands of casitas by eye would be impractical. There are additional image acquisition and advanced analyses options including multiple sensor imagery and pattern recognition that may also prove beneficial.

All in all, there were no detrimental effects of casitas on lobster biology and fishery sustainability detected in the current study. Thus, there is no ecological evidence that the way casitas are used in The Bahamas needs to be altered. The absence of negative effects could also allow fishers to enjoy the enhanced efficiency (including reduced operational costs and reduced use of fossil fuels) that casitas are thought to cause due to the aggregation of lobsters. Nevertheless, the risk of overcapitalization would also have to be considered if more fishers are able to continue to fish due to higher chances of profitability resulting from increased efficiency. Still, efficiency is highly dependent on fishers identifying locations where lobsters are likely to aggregate under casitas and the likelihood that fishers do not remove lobsters from each other’s casitas. These are examples of social, law enforcement, and ethical influences on sustainability. The current study also showed that the use of remote sensing appears possible after further refinement of detection strategies. Meanwhile, the use of wooden lathe traps may put the fishery at risk due to shell disease. Though this risk may be small at this time, it is an unnecessary risk as the law already requires that traps be removed from the sea during the closed season. While it has long been known that traps can have a number of negative impacts on fisheries, the results of the current study reaffirm this possibility and also reaffirm the need to actively monitor and control them.


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VITA

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