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Abundance estimates of the Indo-Pacific lionfish *Pterois volitans/miles* complex in the Western North Atlantic

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Abstract Less than a decade after being observed off Florida, the invasive Indo-Pacific lionfish is now widely distributed off the southeast coast of the United States. As a step towards measuring invasion impacts to native communities, we examine the magnitude and extent of this invasion by first, compiling reports of lionfish to provide range information and second, estimate lionfish abundance from two separate studies. We also estimate native grouper (epinepheline serranids) abundance to better assess and compare lionfish abundances. In the first study we conducted SCUBA diver visual transect surveys at 17 different locations off the North Carolina coast in water depths of 35–50 m. In the second study, we conducted 27 Remote Operated Vehicle (ROV) transect surveys at five locations from Florida to North Carolina in water depths of 50–100 m. In both studies, lionfish were found to be second in abundance only to scamp (*Mycteroperca phenax*). Lionfish were found in higher abundance in the shallower North Carolina SCUBA surveys ($\bar{x} = 21.2\ \mathrm{ha}^{-1}$) than in the deep water ROV surveys ($\bar{x} = 5.2\ \mathrm{ha}^{-1}$). Lionfish reports continue to expand most recently into the Bahamas, raising the specter of further spread into the Caribbean and Gulf of Mexico. The potential impacts of lionfish to native communities are likely to be through direct predation, competition and overcrowding. The high number of lionfish present in the ecosystem increases the potential for cascading impacts throughout the food chain. Within the southeast region the combined effects of climate change, overfishing and invasive species may have irreversible consequences to native communities in this region.

Keywords Lionfish · Invasive marine fish · Serranidae · *Pterois* · Invasion impact · Abundance · Introduced species

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Introduction

Invasive species are altering ecosystems around the world and are increasingly recognized as a major threat to ecosystem health and global biodiversity (Carlton and Geller 1993; Ruiz et al. 1997). Within the past 20 years, there has been an exponential increase in the number of documented marine invasions primarily due to the global transport of marine invertebrates in ballast water (Cohen and Carlton

1998; Carlton 2001). High profile invasions such as the zebra mussel in the Great Lakes (Mills et al. 1993), have raised considerable alarm regarding the detrimental economic and environmental impacts caused by marine invaders (Carlton 2000; Pimentel 2000). In contrast marine fish introductions are still considered relatively rare and historically the environmental effects are often discounted (Randall 1987; Baltz 1991). This is largely due to the intentional nature of most marine fish introductions and difficulties associated with conducting invasion impact studies within open marine environments (Bax et al. 2001).

Most successful marine fish invasions have occurred in closed systems such as inland seas, coastal bays and estuaries (Baltz 1991) or where the invader is anadromous as in the case of Atlantic Salmon (*Salmo salmo*) on the West Coast (Volpe et al. 2000; Volpe 2001). A successful marine fish introduction is the snapper *Lutjanus kasmira* that was intentionally introduced into Hawaiian waters in the 1950s (Randall 1987). Recent studies indicate that *L. kasmira* is now the second most abundant fish both in numbers and biomass over hard substrate in Hawaii (Friedlander et al. 2002).

More recently the Indo-Pacific lionfish (*Pterois volitans/miles* complex)[1] has become established along the southeastern region of the United States. A combination of verified and unverified reports suggest that the lionfish distribution may be continuous from Miami, Florida north to Cape Hatteras, North Carolina (26° N–35° N), in water depths of 30–100 m, including Bermuda (personal communication, Judi Clee, Sarah Manuel), (Whitfield et al. 2002; Hare and Whitfield 2003; Semmens et al. 2004; Ruiz-Carus et al. 2006). Juveniles have also been sporadically reported off the coast of New Jersey, Long Island and Rhode Island, during late summer and fall, but over-wintering survival is not expected due to cold water temperatures (Kimball et al. 2004). These data and observations strongly suggest that lionfish are firmly established, reproducing (Ruiz-Carus et al. 2006) and their population is growing along the Atlantic coast. Nevertheless, this apparent increase in lionfish distribution could be a function of greater public awareness and reporting. Quantitative abundance measures are necessary to understand the true status of the lionfish population and to evaluate potential impacts to native communities.

Measuring impacts of an introduced species in a new environment remains one of the most difficult aspects of invasion biology (Parker et al. 1999; Bax et al. 2001). Often there is no record of the original community before the arrival of the invader and once the invader becomes widespread establishing effective control treatments is difficult (Taylor et al. 1984; Parker et al. 1999). The overall impact of an introduced species has been defined in previous work as a product of the range (distribution) (m^2), abundance (density) and per capita effect of the individual invader (Parker et al. 1999). Therefore our immediate goal is to determine the magnitude and extent of the lionfish invasion, by quantifying lionfish distribution and abundance at two different geographic scales; local (10 s of km) and regional; (100 s of km), thereby, describing 2 of the 3 factors important in assessing potential ecosystem impacts (Parker et al. 1999).

Our objectives for this paper are as follows: (1) present the most recent lionfish distribution data; (2) establish a baseline of population abundance for lionfish to determine the geographic extent and magnitude of the invasion; (3) establish baseline population abundance for groupers, likely competitors of lionfish; and (4) discuss the potential impacts of lionfish, consider their potential for continued dispersal throughout the western Atlantic and identify future research needs in order to directly measure impacts.

Lionfish ecology and biology

The Indo-Pacific lionfish (*Pterois volitans/miles* complex, Scorpaenidae) is a venomous predator (Halstead 1970) native to the sub-tropical and tropical

[1] Analyses of mitochondria-encoded cytochrome-*b* gene sequences have shown that two lionfish taxa, *Pterois volitans* and *P. miles*, are present in the Atlantic (Hamner and Freshwater, personal communication). Analyses of partial cytochrome-*b* and mitochondrial 16S rDNA sequences from multiple species of the lionfish genera *Pterois* and *Dendrochirus* resolved *P. volitans* and *P. miles* as closely related sister taxa, but there is still confusion as to whether they represent two separate species or populations (Kochzius et al. 2003; Hamner and Freshwater personal communication). A morphometric study of specimens from their native range revealed clear differences between these two taxa (Schultz 1986), but examination of the distinguishing characters cited by Schultz (1986) was equivocal for Atlantic specimens. Hereafter, we will be referring to both taxa collectively as lionfish.

regions of the South Pacific, Indian Oceans and the Red Sea (Schultz 1986). Lionfish are generally well known and recognized as a popular aquarium fish. Unfortunately, their ecology and life history characteristics are not well known. Within their native range they are found on coral reefs and rocky outcrops from the surface to 50 m (Schultz 1986). They are opportunistic predators consuming fish, shrimp and crabs (Sano et al. 1984; Fishelson 1997) and are reported to grow to a length of 38 cm (Randall et al. 1997; Myers 1999). Their eggs and larvae are pelagic and are capable of dispersing large distances via ocean currents (Imamura and Yabe 1996). Adults reportedly have few if any predators, likely due to their venomous spines (see Bernadsky and Goulet 1991).

Fig. 1 Group of lionfish found in low relief hard bottom habitat off the North Carolina Coast approximately 150 ft deep. Photo courtesy of Doug Kesling

Methods

Lionfish distribution

Following a National Oceanic and Atmospheric Administration (NOAA) press release in January 2002 (NOAA 2002-R105 http://www.publicaffairs.noaa.gov/releases2002/jan02/noaa02r105.html), NOAA began archiving all lionfish sightings and reports from the general public. Lionfish reports that could not be verified with photograph, video, site location or by an experienced observer were excluded from the archive. The location of these reports were examined to determine the current spatial extent of the invasion (Fig. 1).

Abundance estimates

To quantify lionfish abundance within the western north Atlantic, two separate studies were conducted. The first examined lionfish abundance along the North Carolina continental shelf. The second documented lionfish abundance within potential Marine Protected Areas (MPA) along the outer continental shelf from northern Florida to North Carolina.

North Carolina SCUBA surveys

In August 2004, we conducted SCUBA diver visual transect surveys at 17 rocky reef and wreck sites on the North Carolina continental shelf at depths of 30–45 m. Along the southeast U.S. coast, the warm Gulf Stream supplies heat to coastal waters year round. This warming effect becomes more pronounced in water depths >30 m and allows tropical fish to inhabit rocky reef along the shelf year round (Parker and Dixon 1998). Lionfish and all epinepheline serranids (i.e., groupers) were enumerated. Grouper were included because they are important fishery species in the region (NMFS 2004), are ecologically similar to lionfish (Sano et al. 1984; Naughton and Saloman 1985; Matheson et al. 1986; Fishelson 1997) and are likely competitors (Naughton and Saloman 1985). Locations were selected based on where we expected lionfish to survive the winter (Kimball et al. 2004) and from known locations of rocky-reef habitat. Sites varied considerably in terms of structure, from low relief, patchy, rock bottom to high relief (~5 m) artificial substrate (shipwrecks). One visual transect was completed per site (Sanderson and Solonsky 1986; Samoilys and Carlos 2000; Schmitt et al. 2002). After conducting visual surveys at each site, divers collected lionfish specimens using pole spears; wet weight and total length of each specimen were recorded.

Fish abundance was estimated from total area surveyed and numbers of each species. Transect length varied between 50 and 100 m. Transect width was determined by underwater visibility. The total area surveyed during each dive was calculated from transect length and width. Abundance on each transect was determined as the number of fish observed divided by the transect area. Overall abundance was calculated as the mean of the transect abundances. Significant differences in abundances between species were evaluated using a Kruskal–Wallace non-parametric ANOVA between all species pairs with $\alpha = 0.05$.

Some species of grouper (i.e., gag) may be underestimated due to their aversion to divers, especially in locations that are frequently visited by spearfishers, while other species may be attracted to divers (Chapman et al. 1974). Previous studies comparing diver survey techniques to other census methods found the visual census method underestimated fish abundance by 40–82% (Sale and Douglas 1981; Brock 1982; Stewart and Beukers 2000; Edgar et al. 2004). To correct for possible detectability issues with grouper, we applied a correction factor ($1.92 \times$ abundance) based on the average percent accuracy of 52% estimated for 3 serranid species (Stewart and Beukers 2000). We did not apply a correction factor to lionfish abundances. Lionfish are diver-neutral in their behavior (Kulbicki 1998) and we wanted our estimates of lionfish abundance to be conservative.

We compared our abundance estimates of grouper with catch data obtained from the National Marine Fisheries Service, Southeast Fisheries Science Center Headboat Survey (Jennifer Potts, NOAA, NMFS, Beaufort, North Carolina) to provide an independent evaluation of rank abundance. Headboat survey data were obtained for North Carolina between 1999 and 2003. Catch-per-unit-effort (CPUE) was calculated as the number of fish caught in a given year divided by the number of angler days and averaged over 4 years. Grouper abundance from the visual dive surveys were compared to the CPUE data using linear regression; a significant positive regression provides support for our visual survey data. Lionfish were not found in headboat surveys during the 1999–2003 time period and were not included in this analysis.

Outer-shelf ROV surveys

In May 2004, we conducted 27 deep water Remote Operated Vehicle (ROV) dives within five proposed Marine Protected Areas (MPA)[2] along the continental shelf edge between North Carolina and northern Florida in depths from 46 to 100 m. ROV dive sites were selected based on known locations of rocky reef habitat and shipboard echosounder reconnaissance. Rocky reef habitat varied from patchy low relief (~0.5 m) to high relief (~20–30 m). Video transects were conducted with a Deep Ocean Engineering, Phantom S-2 ROV. All epinepheline grouper and lionfish within a 5 m radius were identified to lowest taxa and counted along each transect. The area of each transect was determined from the transect length (L) and width (W) (Koenig et al. 2005). Transect length was calculated from the latitude and longitude recorded by the ROV tracking system. Width transect was calculated using the following equation:

$$W = 2\left(\tan\left(\frac{1}{2}A\right)\right)(D) \tag{1}$$

where A is the horizontal angle of view (78°, a constant property of the camera) and D is the distance from the camera at which fish could always be identified. The distance (D) was usually 5 m except for two dives where visibility was only 3 m. Transect area (TA) was then calculated as:

$$\mathrm{TA} = (L \times W) - 1/2(W \times D) \tag{2}$$

Abundance of each species was calculated by dividing the number of individuals observed by the TA. The same correction factor used for SCUBA visual transect data was then used for calculating grouper abundance from the ROV data; similarly no correction factor was applied to lionfish abundances. Fish densities for each transect were averaged for each MPA. Data were then pooled across MPAs and significant differences in abundances between species were evaluated using a Kruskal–Wallace non-parametric ANOVA between all species pairs with $\alpha = 0.05$.

[2] Informational Public Hearing Document on Marine Protected Areas to be included in Amendment 14 to the fishery Management plan for the Snapper Grouper Fishery of the South Atlantic Region. South Atlantic Fishery Management Council, January 2004 http://www.safmc.net (last visited Sept 26, 2005).

Results

Lionfish distribution

To date, NOAA has catalogued lionfish sightings from Florida to Rhode Island including Bermuda and most recently the Bahamas (Fig. 2). With only one exception, all lionfish reports occurred within the past 6 years, only reports from West Palm Beach, Florida occurred earlier, in the early to mid 1990's (Hare and Whitfield 2003). Overall these reports represent at least 65 separate locations from the U.S. east coast and Bermuda and 10 different locations within the Bahamas where lionfish have been most recently reported to NOAA (Fig. 2). All lionfish reports north of Cape Hatteras were juveniles (~2.5 cm).

Abundance estimates

North Carolina SCUBA surveys

Lionfish were observed on transects at 13 of the 17 locations (Fig. 3a) and abundances ranged from 0 to 72.7 individuals ha^{-1} ($\bar{x} = 21.2$, std. dev. = 5.1, $n = 17$). Lionfish abundance was not significantly different than the abundance of three grouper species, (*Cephalopholis cruentatus*—graysby, *Mycteroperca microlepis*—gag, *Epinephelus adscensionis*—rock

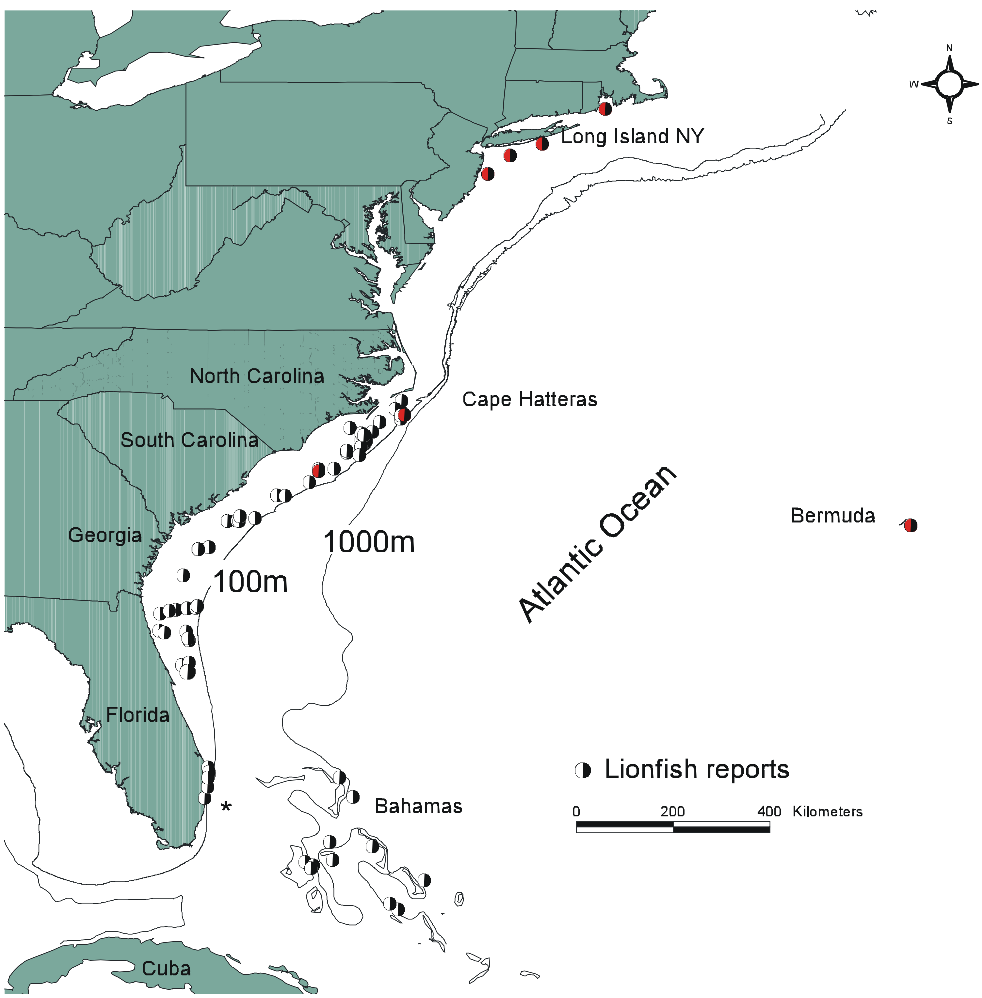
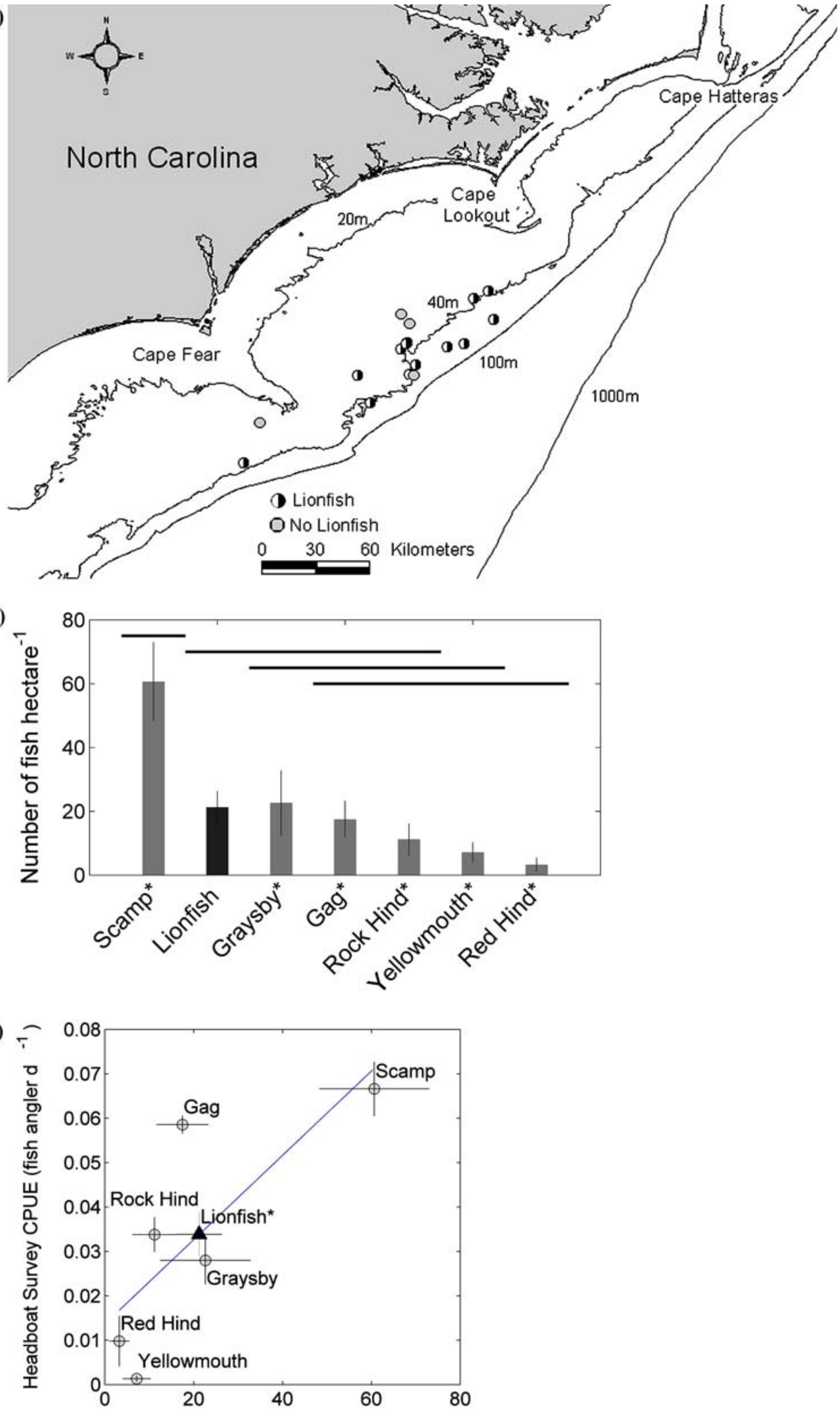


Fig. 2 Map of the East Coast of the United States and the western Atlantic Ocean showing locations of lionfish sightings from August 2000 to January 2005. Sightings of juveniles (<5 cm) are denoted by red/black dots. Sighting of adults (>10 cm) are denoted by white/black dots. *Only location where reports go back to early to mid 1990's

(a)
(b)
(c)


North Carolina
Cape Hatteras
Cape Lookout
Cape Fear
20m
40m
100m
1000m
Lionfish
No Lionfish
0 30 60 Kilometers
N
S
E
W
Number of fish hectare^{-1}
Scamp*
Lionfish
Graysby*
Gag*
Rock Hind*
Yellowmouth*
Red Hind*
Headboat Survey CPUE (fish angler d^{-1})
Visual Survey Abundance (fish hectare^{-1})
Scamp
Gag
Rock Hind
Lionfish*
Graysby
Red Hind
Yellowmouth

◀
Fig. 3 (**a**) Map of diver visual survey locations on the North Carolina continental shelf. Gray circles are transects where no lionfish were observed. Black/white circles are transects where lionfish were observed. (**b**) Mean lionfish and grouper abundances from the diver visual surveys. Error bars represent standard errors based on 17 transects, one conducted at each dive site. Horizontal bars indicate fish densities that were not significantly different. A correction factor (×1.92) applied to grouper densities is indicated by an *. (**c**) Lionfish and grouper abundance from visual dive surveys compared to catch-per-unit-effort (CPUE) data from the NMFS Southeast Fisheries Science Center Headboat Survey. Errors bars for visual census data represent standard errors of mean from 17 transects. Error bars for CPUE represent standard errors based on the 4 years of data. *Lionfish are rarely captured in fisheries but the abundance estimated for headboat CPUE is based on the regression (CPUE=0.011+0.0019*VC; r^2= 0.79) and is shown here for illustrative purposes

hind), were significantly greater than two species (*Mycteroperca interstitialis*—yellowmouth grouper, *Epinephelus guttatus*—red hind) and significantly less than one species (*Mycteroperca phenax*—scamp) (Fig. 3b). Our visual assessment of grouper abundances were significantly correlated to catch-per-unit effort data collected from the headboat fishery in North Carolina (r^2= 0.79) (Fig. 3c), providing independent verification of the rank order of grouper abundances derived from the visual surveys. From these surveys, we conclude that lionfish are now as abundant as many native grouper species.

A total of 149 lionfish were collected. Total length ranged from 5 to 45 cm (average length = 30.5 cm). The average wet weight of the specimens was 480 g and the weights ranged from 25 to 1380 g (3 lbs).

Outer-shelf ROV surveys

Lionfish were observed in all potential MPAs from Florida to North Carolina (Fig. 4a) in water depths ranging between 48 and 84 m. Average lionfish abundance within each potential MPA ranged from 0.62 to 4.6 individuals ha^{-1} (Fig. 4b). Lionfish were the second most abundant species and only lionfish and scamp were observed in all five potential MPAs (Fig. 4b). Combining data from all potential MPAs, lionfish were found in significantly higher abundances than other grouper species (*C. cruentatus*, *M. microlepis*, *E. guttatus*, *Epinephelus morio*—red grouper, *Epinephelus niveatus*—snowy grouper, *Epinephelus drummondhayi*—speckled hind) with one exception: scamp were significantly more abundant than lionfish. These results are in direct agreement with the diver visual transect data, adding further evidence that lionfish population size appears to be similar to or greater than most native grouper species in the ecosystem.

Discussion

The goal of this study was to quantify two of the three components important in determining the overall impact of an introduced species (Parker et al. 1999): the range (distribution) (m^2), abundance (density) and per capita effect of the individual invader (Parker et al. 1999). We show that lionfish are continuously distributed from south Florida to North Carolina and also found in the Bahamas, Bermuda and along the northeast U.S. shelf as juveniles. These data leave little doubt that lionfish are now well established in portions of the western Atlantic Ocean. Abundance estimates from Florida to North Carolina indicate that lionfish are now as abundant as many native grouper species. This is remarkable given the short time period within which this population growth has occurred (Whitfield et al. 2002; Hare and Whitfield 2003). Unfortunately lionfish abundance estimates within their native habitat are lacking and no comparisons can be made. Several lionfish collected in this study were larger (45 cm) than the reported maximum length from their native range (38 cm) (Schultz 1986; Randall et al. 1997; Myers 1999). This finding suggests that lionfish growth along the southeast U.S. is not resource limited (Elton 1958). The rapid establishment of lionfish may be due in part to resource availability that results from over-fishing of potential competitors such as groupers (Davis et al. 2000). An effective larval dispersal mechanism (planktonic eggs and larvae) and generalist diet may have also factored in lionfish success (Ehrlich 1989). There continues to be fishing pressure on native groupers, and limited fishing mortality on lionfish, and thus favorable conditions are likely to continue for lionfish population growth (Moulton and Pimm 1986). Thus, we conclude that the distribution and abundance are likely to increase further and that the impact of lionfish on the ecosystem will also continue to increase.

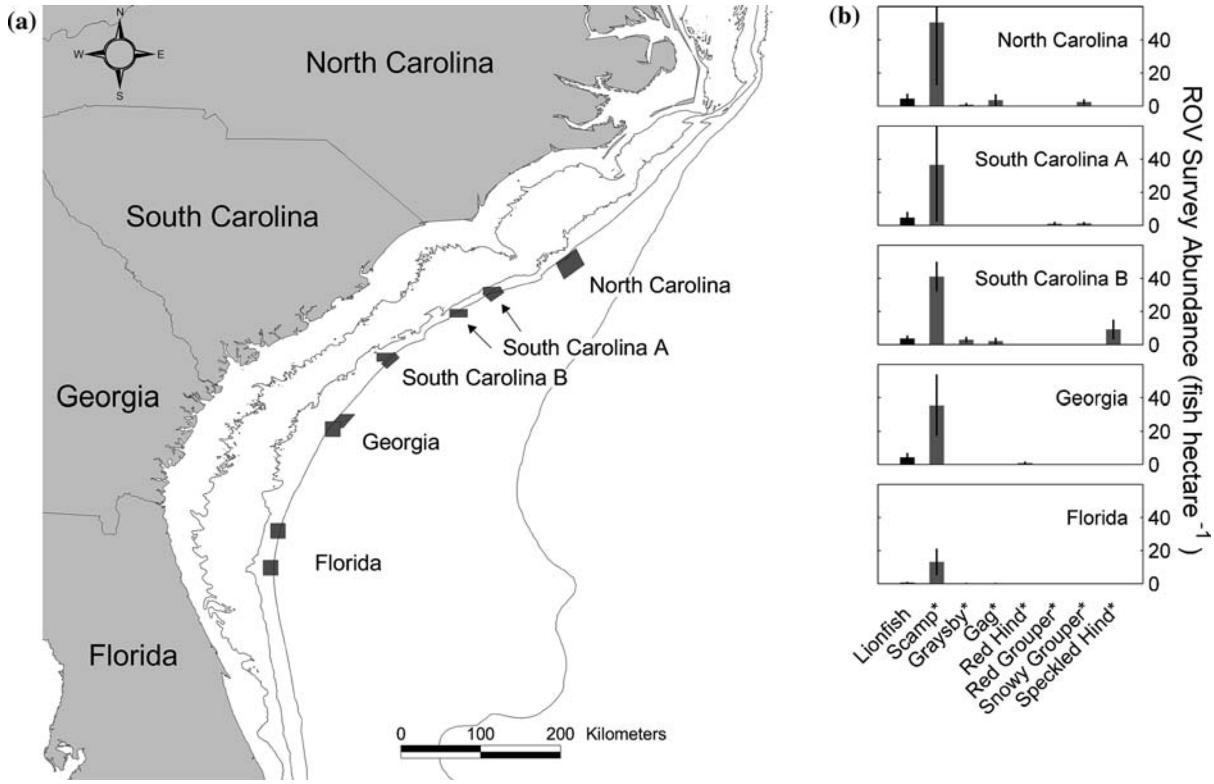


Fig. 4 (**a**) Map of five MPAs proposed by the South Atlantic Fishery Management Council, Amendment 14, ranging from North Carolina to the Florida Keys. The proposed MPAs surveyed include: North Carolina (2 Options), South Carolina A (3 Options), South Carolina B (2 Options), Georgia (2 Options), and Florida (2 Options). (**b**) Mean lionfish and grouper abundances for each proposed MPA. *Correction factor applied to Grouper densities

Dispersal into the Caribbean

A major concern raised by the distribution data presented in this study (Fig. 2), is the potential spread of lionfish into the Caribbean and Gulf of Mexico. The spatial pattern and timing of the lionfish reports suggests Florida as a likely introduction source of the founding population (Fig. 2) (Whitfield et al. 2002; Hare and Whitfield 2003; Ruiz-Carus et al. 2006). Since 2005, lionfish have been reported in the Bahamas; these reports come 5 years after the first report off the coast of North Carolina. It is unknown whether these fish were naturally dispersed from the southeast U.S. or represent new local introductions; either way the occurrence of lionfish in the Bahamas raises the specter of further spread into the Caribbean and Gulf of Mexico. Minimum winter bottom water temperature is an important factor in controlling the distribution of lionfish within its introduced range, but minimum water temperatures in the Gulf of Mexico and throughout the Caribbean are above documented thermal limits (Kimball et al. 2004).

Although connectivity from the Bahamas and southeast U.S. into the Caribbean and Gulf of Mexico may be very low (Cowen et al. 2006), there may be specific regions (e.g., the Turks and Caicos Islands) that have higher connectivity with the rest of the Caribbean (see Fig. 4 in Cowen et al. (2006)). Preliminary genetics data from lionfish specimens collected in North Carolina suggest that a small founding population of no less than three females may be responsible for the entire population (Hamner and Freshwater unpublished data). If such a small number of individuals could be the founders for the population—a population that exceeds many native groupers in numbers—even minimal larval connectivity from the southeast U.S. and Bahamas could lead to invasion of the Caribbean and the Gulf of Mexico through a stepping-stone effect (Carr and Reed 1993; Cowen et al. 2006). Increased monitoring efforts are needed and targeted eradication efforts should be considered by national and regional management agencies; however, given the reported range and the number of individuals, eradication efforts will need to be focused on critical dispersal chokepoints (see Hare and Whitfield 2003). These chokepoints could be defined through modeling efforts such as those conducted by Cowen et al. (2006).

Potential impacts

Introduced fishes in aquatic habitats can impact native ecosystems in a variety of ways. For clarity we define ecological impacts based on Taylor et al. (1984):

> ''any effect attributable to the exotic that causes – either directly or indirectly – changes in the density, distribution, growth characteristics, condition, or behavior of one or more populations within that community''.

These impacts have been grouped into five general categories; (1) habitat alterations, (2) introduction of parasites or diseases, (3) trophic alterations, (4) hybridization, or (5) spatial alterations (Taylor et al. 1984).

Ecosystem effects from introduced fishes within freshwater environments have been demonstrated in each of these categories and have resulted in the decline and displacement of native fish populations (Taylor et al. 1984; Marchetti 1999; Godinho and Ferreira 2000; Stapp and Hayward 2002; Taniguchi et al. 2002) and other community level effects that extend beyond the replacement of native fishes (Taylor et al. 1984; Miller et al. 1989; Flecker and Townsend 1994; Jude et al. 1995; Englund 1999; Godinho and Ferreira 2000; Stapp and Hayward 2002). But there has been very little research that directly examines the impact of a marine fish in an open marine system (but see Friedlander et al. 2002; Morales-Nin and Ralston 1990).

One example of a marine fish invader for which some impacts have been examined is the snapper *L. kasmira* in Hawaii. Age and growth studies indicate a faster growth rate of *L. kasmira*, especially when compared with Atlantic lutjanids (Manooch 1987) and native Hawaiian deep-water species (Morales-Nin and Ralston 1990). *Lutjanus kasmira* may be benefiting from increased resource availability, due to a lack of competitors in the near shore Hawaiian waters (Morales-Nin and Ralston 1990). Other than direct predation, no other community interactions or impacts by *L. kasmira* have been examined or discussed (Oda and Parrish 1981).

Since lionfish are opportunistic predators feeding primarily on smaller fishes, there is considerable potential for ecological overlap with native fish species (Sano et al. 1984; Naughton and Saloman 1985; Matheson et al. 1986; Fishelson 1997). Continued mortality of groupers and other native predators through overfishing (Huntsman et al. 1999; NMFS 2004) may open niche space and further increase resources for lionfish (Davis et al. 2000). In contrast to many native fishery species, lionfish experience little fishing mortality and potentially lower natural mortality owing to venomous spines, this combination is likely to give lionfish a competitive advantage over native species such as grouper; such benefits may explain the rapid increase in population abundance from first reports in as little as 5 years. Lionfish may also affect the use of habitat by other species through physical overcrowding and aggressive tendencies. This threat is likely to increase as lionfish abundance increases and may cause native species displacement to sub-optimum habitats (Taylor et al. 1984). Lionfish if not aggressive are often described as 'standing their ground' and exhibiting 'no fear' of divers (Myers 1991) a behavioral characteristic that may also extend to native fishes. The high number of lionfish now present in the ecosystem increases the potential for cascading impacts throughout the food chain. However, the magnitude of impacts within an open marine system remain unknown.

Measuring impacts

Quantitative impact studies are generally lacking within estuarine and marine invasion literature, even in the most invaded estuaries (Cohen and Carlton 1998; Parker et al. 1999; Ruiz et al. 1999; Bax et al. 2001). In addition, no standardized frame work or rules for measuring impact exist in the literature (Parker et al. 1999). In this paper, we have described two of three factors important in determining the overall lionfish impact but the third, and most difficult factor remains, the per-capita or per-biomass effect (E) (Parker et al. 1999). Measuring invader impacts can only be determined through quantitative experimental impact studies that are adequately controlled to minimize confounding environmental factors. Often these controlled studies are not possible, but in their absence, invader impact studies primarily utilize two approaches; (1) before and after impact comparisons and (2) a correlation between an invader and a change in the native community (Taylor et al. 1984). Both approaches yield little explanation of the causal mechanism behind invader impact, but can be a good starting point for experimental work. In the case of lionfish they have already established a broad geographic and depth range, where there is little if any quantitative baseline data characterizing the original community. Therefore in most cases, direct comparisons between community data before and after lionfish arrived will not be possible. Further, rigorous and repeated removal of lionfish will be necessary to establish control treatments. This of course may not be entirely successful but a local decrease in population may be achieved and lionfish recruitment over time could then be examined.

Once these obstacles are overcome, lionfish impacts could potentially be measured by combining quantitative in-situ studies with controlled laboratory experiments. Ideally field studies should include a variety of metrics (i.e. density, diversity indices, etc.), levels of organization (individuals, populations etc.) and if possible multiple spatial scales (Parker et al. 1999). The laboratory experiments should incorporate both native community and lionfish interactions (i.e. competition, predation, etc.) (Taylor et al. 1984; Parker et al. 1999). Of course, rarely do researchers have all the resources they need to conduct such thorough invader impact studies (Bax et al. 2001). A more directed approach would be to focus impact studies on native species that are most likely to be affected by lionfish such as prey species or potential competitors of lionfish (for food or habitat). A thorough examination of lionfish life history characteristics should also be undertaken and several studies are currently underway (Munoz et al. unpublished data; Potts et al. unpublished data). Other research needs include tagging individual lionfish to increase our understanding of movement patterns and in-situ growth rates. Tagging at the observed inshore limit (<30 m) could yield additional information regarding lionfish over-wintering survival at their thermal limits, to resolve the actual inshore depth limit for over-wintering along the southeast shelf.

Even under ideal conditions, conducting controlled field experiments that test a specific hypothesis are difficult. Within an open marine system, the sheer amount of variation within the ecosystem further complicates issues of experimental design, choice of scale, metric and isolation of variables to examine invader impacts (Taylor et al. 1984; Parker et al. 1999; Ruiz et al. 1999). These experimental issues are compounded further by logistic difficulties, especially in water depths >30 m, where there are serious limitations associated with research time, working in remote locations, trained personnel, weather, and research vessel costs.

Conclusion

Within the western north Atlantic many predatory fish species in the ecosystem are over-fished (Huntsman et al. 1999; NMFS 2004). Off the North Carolina coast there has already been a documented shift in faunal composition, from temperate to tropical species associated with a 1°C rise in winter bottom water temperatures (Parker and Dixon 1998). In addition to lionfish, 14 other Pacific marine fish species are currently surviving off the coast of Florida (Semmens et al. 2004). One being a predatory grouper, *Cromileptes altivelis* with high potential to become established. The effect of climate change, overfishing and invasive species have been implicated in ecosystem decline and collapse in several marine ecosystems (Harris and Tyrrell 2001; Stachowicz et al. 2002; Frank et al. 2005). Along the southeast U.S. shelf the high number of stressors acting in synergism may eventually have unexpected and irreversible consequences for the native communities and economically valuable fisheries in this region. This scenario implies a direct economic cost within an open marine environment that is related to invasive species—a cost which is just beginning to be recognized.

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