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Research Article

The role of conservation volunteers in the detection, monitoring and management of invasive alien lionfish

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Abstract

Across the Caribbean, targeted fishing is gaining momentum as a cost-effective method to control invasive alien lionfish (*Pterois volitans* and *Pterois miles*) by suppressing population numbers below site-specific threshold levels i.e. a population density that is predicted to cause declines in native fish biomass. Yet in marine reserve no take zones (NTZs) or reefs at depths of >18 m) where commercial fishing is either not permitted or impractical, alternative methods of lionfish control are required. This study evaluates the potential for conservation volunteers to act as citizen scientists monitoring invasive lionfish populations and to support removal efforts in Bacalar Chico Marine Reserve (BCMR), Belize. Two underwater visual census techniques were trialled with conservation volunteers, each with associated benefits and drawbacks. A log of opportunistic lionfish sightings on SCUBA dives has been used to record sightings per unit effort (SPUE) data 2011–2015. In 2014, more rigorous lionfish focused searches (lionfish-adapted belt transects) were introduced. Opportunistic lionfish sightings contributed to a five year SPUE dataset that suggests that lionfish population growth rate has slowed in BCMR, where a lionfish removal program was also carried out by conservation volunteers over the same timeframe. However, lionfish focused searches showed that the mean density in 2014 was high (mean = 27.05 ± 8.77 fish ha^{-1}, 1–30 m) relative to lionfish populations in their native ranges, particularly at sites at depths > 18m (mean = 43.39 ± 13.76 fish ha^{-1}, 18–30 m). Drawing on lessons from Belize, we discuss the potential for conservation volunteers to support invasive alien species (IAS) monitoring and control efforts in marine environments.

Key words: non-native species, citizen science, *Pterois volitans*, marine invasions, Caribbean, fisheries

Introduction

Citizen science—a term describing collaborations between volunteers and scientists to answer research questions (Crall et al. 2010)—directly supports conservation by contributing data and manpower for monitoring and evaluation (Dickinson et al. 2010). It can also increase participants' scientific literacy and boost environmental awareness (Bonney et al. 2016; Vitone et al. 2016).

Over the past decade, there has been a particular rise in the use of citizen science in the recording of invasive alien species (IAS), with members of the public able to monitor IAS across far wider spatial and temporal scales than would be economically or logistically feasible by professional researchers alone (Delaney et al. 2008; Crall et al. 2013; Scyphers et al. 2015; Hyder et al. 2015). Such studies have allowed species to be detected earlier, and range expansions observed over far wider spatial scales, both of which are essential for rapid and effective management (Delaney et al. 2008; Scyphers et al. 2015).

Historically, the majority of citizen science initiatives has taken place in terrestrial ecosystems (Tweddle et al. 2012). This is somewhat inevitable given the expense, technical skill and equipment needs (SCUBA diving) associated with visiting marine environments, immediately limiting the potential pool of participants (Cigliano et al. 2015). However volunteer SCUBA divers are increasingly supporting efforts to address marine conservation issues (Hyder et al. 2015; Cigliano et al. 2015). For example the number of voluntary surveys conducted as part of the international REEF Fish Survey Project grew from 6,700 surveys in 2000 to 11,222 in 2015 (Reef Environmental Education Foundation 2015). With an estimated six million SCUBA divers worldwide (The Diving Equipment and Marketing Association 2013) there is strong potential for more divers to support professional researchers and managers with marine monitoring efforts (Ward-Paige and Lotze 2011).

Recreational SCUBA divers can offer cost-effective and reliable assistance with coral reef monitoring and have recorded reef sharks and fish, which are often easy to observe and identify (Azzurro et al. 2013), with the same level of accuracy as professional scientists (Mumby et al. 1995; Goffredo et al. 2010; Ward-Paige and Lotze 2011; Holt et al. 2013; Branchini et al. 2015).

The data recorded by recreational divers have helped to inform policy changes on the location of marine protected areas yet their potential to assist with monitoring the distribution of marine IAS, which are often patchily distributed, is considered particularly valuable (Hyder et al. 2015). Citizen science initiatives have already shown promise at recording marine alien IAS. In Florida, fishers and recreational divers documented the lionfish invasion 1–2 years earlier than tradition coral reef monitoring programs (Scyphers et al. 2015). In the Mediterranean Sea, recreational divers taking part in the "Sea-watchers" citizen science project were the first to detect the Sergeant major *Abudefduf saxatilis* (Linnaeus, 1758), a species of damsel fish native to the tropical Atlantic (Azzurro et al. 2013). While in Greece, recreational divers, underwater photographers and fishers together recorded 28 alien marine species (later validated by taxonomic experts) in 2012 (Zenetos et al. 2013).

Conservation volunteers, those who volunteer to undertake holidays that might involve the restoration or research of certain environments (Wearing 2001), offer a self-funding source of assistance with ecological monitoring, particularly valuable to countries with financial or human resource constraints (Mumby et al. 1995).

Lionfish

The invasion of lionfish (*Pterois volitans* and to a lesser extent *P. miles*) in the Caribbean is one of the best documented examples of a marine IAS (Côté et al. 2013; Scyphers et al. 2015). Native to the tropical Indo-Pacific, lionfish were first reported in the Atlantic Ocean off South Florida in the 1980s, with release from aquaria the most likely introduction route (Morris and Whitfield 2009; Semmens et al. 2004). The species quickly established and spread, and its introduced range has now expanded as far north as New York (Schofield et al. 2017) and as far south as Brazil (Ferreira et al. 2015).

With a prey consumption rate three times that of native Caribbean counterparts such as the coney grouper (*Cephalopholis fulva* Linnaeus, 1758) (Albins 2012) and the capacity to release up to two million eggs per year (Morris and Whitfield 2009)—compared to 300,000 eggs per year by native mesopredators (Côté et al. 2013)—the lionfish invasion can cause reductions in native fish biomass (Green et al. 2012a) and diversity (Albins 2012). It also threatens to cause the competitive exclusion of other native predators (Morris and Whitfield 2009; Green et al. 2011; Albins 2012), potentially threatening the region's small-scale fisheries (Arias-González et al. 2011; Chagaris et al. 2015).

The eradication of lionfish is no longer considered possible (Barbour et al. 2011), however reducing lionfish populations to below predicted site-specific threshold densities relative to native fish communities could lead to an increase in native fish biomass (Green et al. 2014). Accurate measures of lionfish abundance and detailed assessments of the native fish community are required to quantify such densities (Green et al. 2012b) and the associated removals are believed to be most effective when conducted on a regular (monthly) basis (Johnston and Purkis 2015), an unrealistic option for resource managers working in isolation. Although the development of a commercial lionfish market may be a feasible management option (Chapman et al. 2016) it is still in its early stages. Moreover, commercial fishing would not allow the removal of lionfish without approved equipment and/or permits in areas inaccessible to fishing, such as no take zones and deeper reefs.

In 2011, lionfish monitoring and removals began in Bacalar Chico Marine Reserve (BCMR), northern Belize with the help of Blue Ventures marine conservation volunteers as citizen scientists. Lionfish are a particularly suitable study species for citizen scientists to work with because their conspicuous morphology means the chance of misidentification by volunteers is low (Scyphers et al. 2015), while the extra manpower available to remove lionfish from areas off limits to fishing due to their depth, or location within no-take zones, makes population suppression an achievable management goal for BCMR.

The aim of this study is to share the results and key lessons learned from the first five years of a citizen science program for the monitoring and removal of invasive lionfish by conservation volunteers in Belize, and to explore the relative merits and drawbacks of two different citizen science monitoring approaches.

Methods

Study site

Bacalar Chico Marine Reserve (BCMR) in northern Belize borders the Mexican protected area *Arrecifes de Xcalak* and is located in the far north of Ambergris Caye (18º08′28″N; 87º51′47″W). The reserve is one of seven Marine Protected Areas (MPA) that collectively comprise the Belize Barrier Reef Reserve System UNESCO World Heritage Site, and is part of the Mesoamerican Barrier Reef system, a biodiversity hotspot (Vásquez-Yeomans et al. 2011). BCMR is a multiple-use MPA with four types of management zone: Preservation Zone (PZ), completely protected, no commercial or recreational activities permitted; Conservation Zone 1 (CZ1), all forms of fishing banned but recreational activities (snorkelling and SCUBA) allowed; Conservation Zone 2 (CZ2), no-take but catch-and-release sport fishing and other recreational activities (snorkelling and SCUBA) allowed; General Use Zones 1 and 2 (GUZ), extractive fishing permitted with a license and gear restrictions (no gill nets or long lines) in place (Grimshaw and Paz 2004). Coral reef habitat within the study area can be loosely classified as backreef (behind the barrier, within the reef lagoon; sheltered) or forereef (outside the barrier; exposed). The forereef comprises a continuous reef crest at 5–10 m depth descending to 30–40 m depth in highly rugose spur-and-groove formations, and the backreef comprises of broken patch reefs at depths of 3–4 m, surrounded by dense seagrass beds.

Belize's first official report of lionfish was from Turneffe Atoll in December 2008 (Searle et al. 2012) and it was first reported in BCMR in March 2010 (Ateweberhan et al. 2011). Since then, multi-sector collaborations between marine conservation organisations (including Blue Ventures, ECOMAR and ReefCI), government agencies (Belize Fisheries Department), fishing cooperatives and fishermen associations have been established to address lionfish population monitoring, research and management (Chapman et al. 2016). The UK marine conservation organisation Blue Ventures has led efforts to monitor and manage populations of lionfish in BCMR, enlisting the help of international conservation volunteers who pay to receive training on and assist with coral reef conservation activities.

Volunteer training and skill verification

Although the level of SCUBA diving experience varied between volunteers, they were all required to have completed PADI Advanced Open Water or equivalent other diving certification to take part in lionfish activities. Before contributing to data collection, volunteers received instruction on how to identify lionfish and their microhabitat, completed in-water training for estimating fish sizes underwater (Bell et al. 1985), and performed one or more practice surveys evaluated by a scientific member of staff.

Method 1: Opportunistic lionfish sightings (2011–2015)

To record opportunistic lionfish sightings, volunteers logged all lionfish they encountered between entering and exiting the water during all types of dive (training, coral reef survey, lionfish cull or fun dive), each lasting approximately 45 minutes and followed a normal recreational dive path (i.e. not a pre-defined transect). Divers were encouraged to look for lionfish in the water column, as well as in the crevices, caves and overhangs that they swam past on all dives, however they were only actively searching for lionfish (i.e. actively searching typical lionfish habitats for extended periods of time) during lionfish culls. When lionfish were sighted, data were collected on: total length (estimated to closest 1 cm, from nose to tip of tail); depth; microhabitat (cave/crevice/overhang, gorgonian bed, seagrass, open reef or sand) and whether the lionfish was culled.

Sightings were recorded on underwater slates during the dive, immediately transferred from slates to standardised paper data sheets upon return from the field site, and double-entered by volunteers into a Microsoft Excel data form at a later stage. Data were verified by a staff member using a macro to highlight discrepancies between the two datasets. When discrepancies were found, these were corrected by referencing paper data sheets.

Sightings per unit effort (SPUE) was calculated by dividing the number of lionfish counted on a dive by the total number of diver hours on that same dive (sum of every diver's actual bottom time in minutes/60). For example, if four lionfish were observed on a five-person dive totalling 3.75 diver hours (45 mins for each diver), the SPUE would be 1.33 ind. diver hour $^{-1}$.

Method 2: Lionfish focused searches (adapted belt transects) (2014 and 2015)

In 2014, dedicated lionfish survey transects were piloted as a more standardised approach to monitor the presence/absence and density of lionfish at different depths, and in different management zones within the BCMR. Conventional underwater visual census methods (belt transects, stationary visual census) have a low probability of detecting lionfish due to the crepuscular activity patterns and cryptic nature of lionfish (Green et al. 2011). The lionfish-focused search (LFS) method (Green 2012) was therefore used, adapted (as detailed below) to enable surveying by SCUBA on spur-and-groove formation coral reefs from 1 to 30 m deep.

At the beginning of a survey, a 30 m transect tape was laid out ensuring a distance of at least 10 m from other transect tapes at the site. Tapes were laid along the reef flats; ± 1 m parallel to a wall; or following a depth contour along a reef slope depending on the area of reef being surveyed.

Volunteers swam a U-shaped search pattern to thoroughly search for lionfish 2.5 m either side of the transect tape (i.e. an area of 30 m × 5 m; 150 m^2) at a maximum speed of 10 m min^{-1} exploring all potential lionfish habitats along the transect (in crevices, under overhangs, around coral heads etc.). When a lionfish was sighted, its position (distance along the transect) was recorded along with depth, total length and microhabitat. To ensure the results were comparable with other studies, lionfish ha^{-1} was calculated by dividing the number of lionfish sighted on one 150 m^2 transect by 0.015 to scale the result up to a one hectare (10,000 m^2) area. Eight dive sites that represented the range of reef types and depths found within BCMR (backreef, 1–5 m; shallow forereef, 8–15 m; deep forereef, 18–30 m) were selected for the pilot study in 2014, ensuring representation of all management zones (see Figure 5). A minimum of six LFS transects were completed by conservation volunteers at each dive site, except where the site was a small patch reef, in which case the maximum number of transects that could be fit on the site were performed. No sites were surveyed within 3 months of a cull and all surveys were conducted between 09.00 and 16.00 to avoid changes in fish behaviour during crepuscular activity periods (Green et al. 2011).

Lionfish culls

Volunteers began supporting lionfish removal efforts in BCMR in March 2011. Since then, removals have taken place on a weekly basis (when weather conditions allow and volunteers are in attendance) and sites have been rotated to spread effort across the reserve, as well as avoid removing any lionfish three months prior to an LFS survey.

The red lionfish is the only member of its subfamily (Pteroinae) and is morphologically distinct from all other species of fish found on Belizean reefs. Prior to assisting with lionfish cull surveys, all divers were trained in lionfish identification and survey methods, as well as in the use of the lionfish culling device, a selective fishing gear (three-pronged Caribbean sling). Training in the use of the device included aim practice on land and in-water on seagrass beds, using coconuts as dummy targets. Care was taken during culls to avoid damage to corals or other benthic organisms.

Typically, groups of 5–6 divers took part in each removal dive. After descending to the planned depth, buddy pairs fanned out to cover a large area of the reef, searching under overhangs and in crevices. When a lionfish was sighted, data were recorded on lionfish size, behaviour and microhabitat. The three-pronged sling was used to quickly spear the lionfish, which was then pushed into a sealed container to prevent injuries.

Historically, the locations of removal dives have been guided by the SPUE results, and sites with the highest SPUE prioritised.

Data analysis

Univariate statistical analyses were performed in R version 3.3.0 (R Core Team 2015). Analysis of Variance (ANOVA) was used to identify significant differences in the size and abundance of lionfish over space and time. Where significant results were found, Tukey's post-hoc LSD tests were performed to determine which pairs of dive types, or years, differed significant from one another. Student's T-tests were performed to determine whether the size structure of lionfish estimated in-water differed from the size structure of culled lionfish.

Results

Opportunistic lionfish sightings

Between 2011 and 2015, conservation volunteers recorded lionfish sightings on 1390 dives. ANOVAs revealed a significant difference in SPUE between years ($F_{4,1384} = 4.13$, $p < 0.001$; Figure 1A), yet posthoc Tukey HSD tests revealed the only pair of years with a significant difference between them was 2012 and 2013 ($p < 0.05$), i.e. before and after the highest intensity of lionfish culls (Figure 1B).

SPUE also varied significantly between types of dive ($F_{5,1332} = 71.69$, $p < 0.001$). Posthoc Tukey HSD tests revealed that SPUE was significantly ($p < 0.05$) higher on lionfish cull dives, during which volunteers were actively searching for lionfish, than all other types of dive (Figure 2).

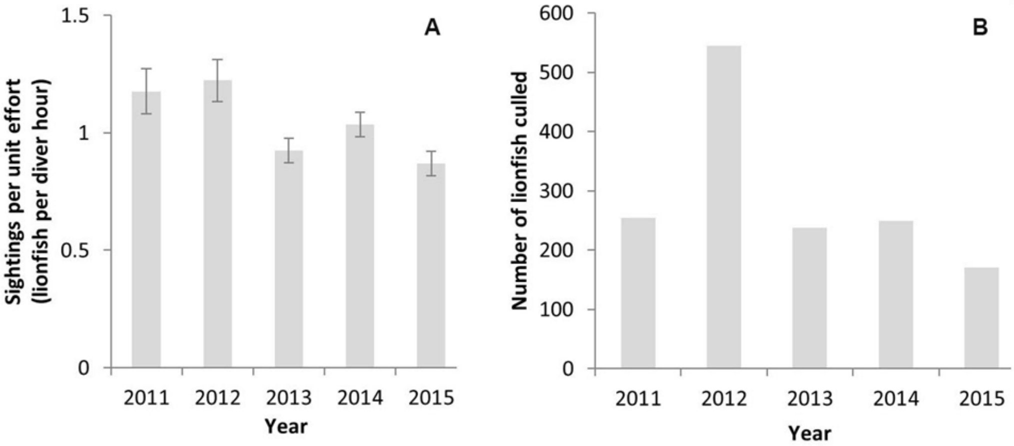

Figure 1. A) Mean sightings per unit effort (SPUE) across all dives since 2011 (error bars show standard error); B) total number of lionfish culled by conservation volunteers since 2011.

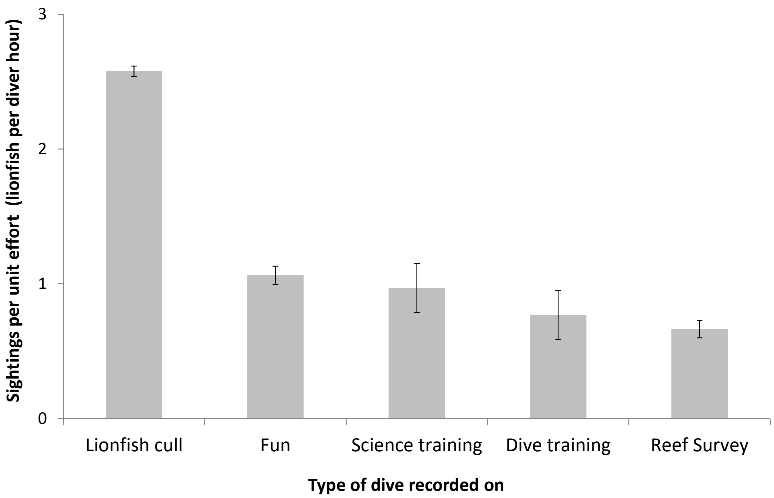

Figure 2. Mean SPUE recorded on different types of dive undertaken by conservation volunteers between 2011 and 2015. Errors bars show Standard Error. Number of dives under each category: lionfish cull n = 136; fun dives n = 127; science training n = 593; dive training n = 275; reef survey n = 220.

A Student's t-test revealed that the size structure of culled lionfish recorded on land did not differ significantly from the size structure of lionfish recorded through in-water length estimates by volunteers ($t = -1.99$, $df = 14$, $p = 0.07$). Individuals sized in-water were therefore considered accurately estimated.

The mean size of lionfish sighted opportunistically on dives increased from 21.8 ± 0.2 cm (n = 876) in 2011 to 23.4 ± 0.3 cm (n = 718) in 2015, peaking at 26.2 ± 0.2 cm in 2014 (Figure 3). ANOVA revealed a significant difference in the size of lionfish between years (ANOVA; $F_{4,5523} = 60.64$, $p < 0.001$).

Lionfish focused searches

In 2014, mean lionfish density in BCMR on coral reefs 1–30 m deep was 27 ± 9 lionfish ha^{-1} (range 0–267 lionfish ha^{-1}). Density varied by depth with the highest density of lionfish recorded on transects at depths of 18–30 m (mean density 40 ± 14 lionfish ha^{-1}) and the lowest density on shallow transects (1–5 m depth; mean density 6 ± 6 lionfish ha^{-1}) (Figure 4).

Lionfish removals

A total of 1455 lionfish removals took place between 2011 and 2015 across a broad spatial scale and all types of management zones in BCMR (Figure 1B and Figure 5). No non-target fish were caught as by-catch during any culls.

Where lionfish were sighted on cull dives, between 1 and 33 individuals were removed per dive (mean = 2.58 ind. diver $hour^{-1}$). The highest numbers of lionfish were removed from sites within Conservation Zones where commercial fishing is banned (Figure 5).

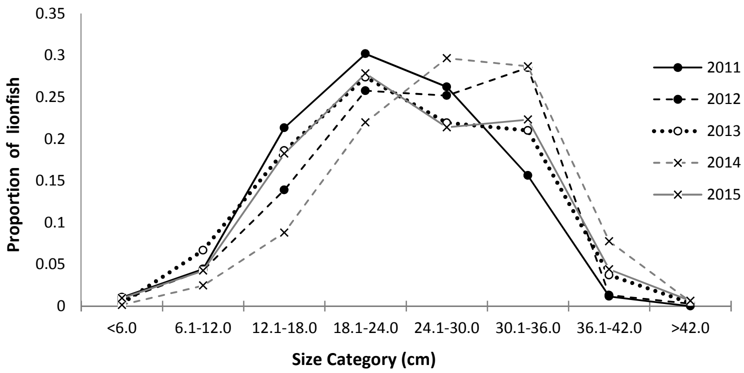

Figure 3. Proportion of lionfish belonging to different size classes for total length (mm), estimated by conservation volunteers while SCUBA diving in BCMR between 2011 and 2015.

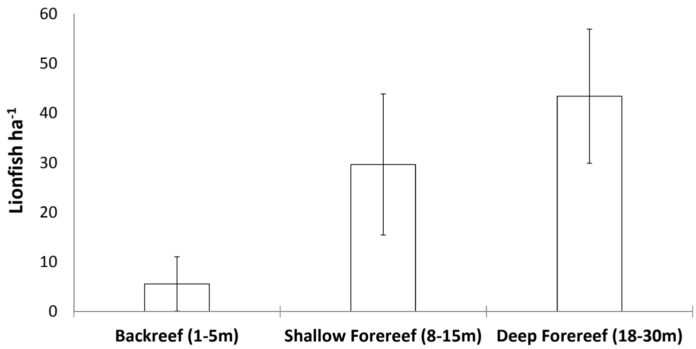

Figure 4. Lionfish density recorded on lionfish focused search pilot surveys in BCMR in 2014, presented by reef type and depth. Errors bars show Standard Error. Number of transects: backreef n = 10; shallow forereef n = 21; deep forereef n = 28.

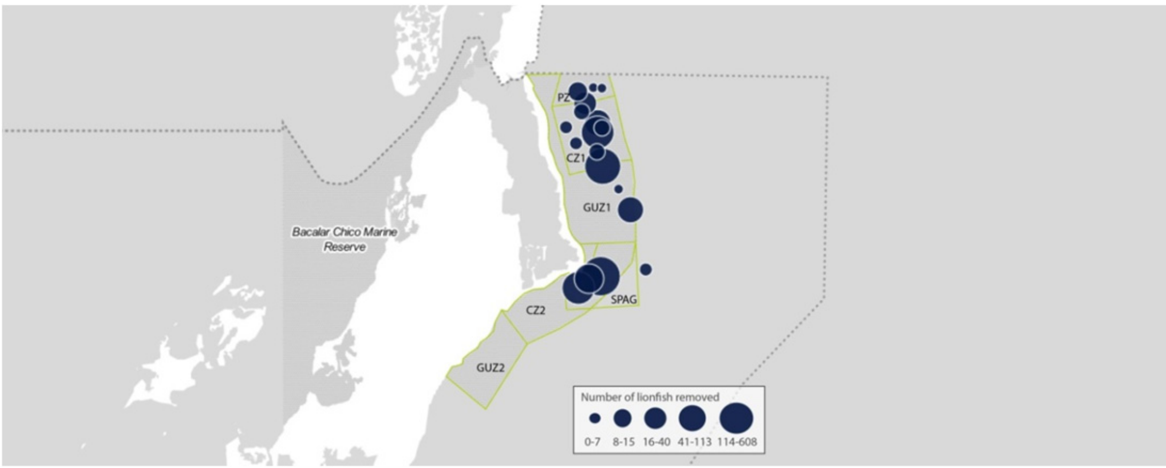

Figure 5. Location and number of lionfish removals between 2011 and 2015 across different management zones within Bacalar Chico Marine Reserve. GUZ1 and GUZ2 = General Use Zones 1 and 2; CZ1 and CZ2 = Conservation Zones 1 and 2; NTZ = No Take Zone; PZ = Preservation Zone.

Discussion

The lionfish program in Bacalar Chico Marine Reserve demonstrates the value of conservation volunteers in monitoring, and assisting with the selective removal of, IAS in marine habitats. Our findings reinforce those of previous studies showing the potential for volunteer SCUBA divers to assist in the monitoring of species in marine habitats at far wider spatial scales than would be possible by professional scientists or resource managers (Mumby et al. 1995; Goffredo et al. 2010; Hyder et al. 2015). Moreover, the fees paid by conservation volunteers helped to sustain funding for the removal program, reducing reliance on government resources or grants.

The LFS surveys, conducted by volunteers in 2014, revealed that the highest densities of lionfish were in sites deeper than 18 m. These results are consistent with a reconstruction of the lionfish invasion in Turks and Caicos which showed lionfish colonised deeper habitats, and that their densities in deeper habitats (10–30 m) surpassed those in shallower habitats (< 5 m deep) as the invasion progressed (Claydon et al. 2012). The recorded increase in lionfish body size (associated with higher egg production) identified through five years of opportunistic sightings also points to the fact that the lionfish invasion is continuing to advance in the BCMR (Claydon et al. 2012), yet the lack of a significant increase in SPUE between 2011 and 2015 may suggest that culling efforts have stemmed the growth of lionfish populations to some extent, something that is difficult to quantitatively assess in the absence of an unmanaged control site.

The fact that the highest densities of lionfish were found in deeper sites poses a challenge to lionfish removal efforts. SCUBA equipment is required to descend to such depths and the duration of dives at depths of 30 m are short, limiting the time available for removals to as little as 20 minutes. Unlike commercial fishers, conservation volunteers are able to safely access these sites with SCUBA gear to conduct lionfish removals, however it remains to be seen whether sufficiently high removal rates would be feasible at such depths. Moreover, lionfish are thought to become more wary of divers if they are unsuccessfully targeted by volunteers, limiting the success of future removal encounters (Côté et al. 2014). Removal rates of at least 30 percent of a lionfish population, repeated at regular intervals rather than as a one-off high intensity event (Morris et al. 2010), are thought to be required to trigger declines in lionfish population size (Morris et al. 2010; Barbour et al. 2011; Scyphers et al. 2015). Insufficient removals may cause rapid population recovery (REEF 2012; Green et al. 2013). Once site-specific threshold levels for lionfish removals have been calculated, the practicality of conservation volunteers achieving the necessary removal efforts at depth will need to be evaluated. Complementary removal strategies such as lionfish derbies (tournaments during which teams of recreational divers compete to remove as many lionfish as possible (Pitt and Trott 2015), may need to be implemented reach the necessary removal targets. Given that lionfish have been recorded at depths of up to 300 metres (Bird et al. 2014), the introduction of lionfish traps (although some have been associated with lobster bycatch (Goffredo et al. 2010; Branchini et al. 2015) may also be needed to remove lionfish from sites deeper than 30m which are inaccessible to recreational divers.

The distinctive morphology of lionfish, and the selective method of removal (spearing individuals one at time) allows highly selective removals to take place. However, we recognise that less conspicuous marine IAS may be more difficult for conservation volunteers to identify and selectively remove. In such situations, further training and validation processes will be required to prevent the unintended capture of non-target species.

Comparing the two monitoring methodologies

The two methods for lionfish monitoring by conservation volunteers each had merits and pitfalls. Opportunistic lionfish monitoring required little volunteer training but was subject to spatial clustering (Scyphers et al. 2015). For example, dive sites were frequently selected based on their suitability for training or weather conditions, an issue that has frequently been identified in other recreational SCUBA-based citizen science initiatives (Delaney et al. 2008). Despite this, the opportunistic sightings results were valuable in identifying broad-scale long-term patterns in SPUE which would still have merit in monitoring responses to management decisions, in monitoring lionfish across larger areas of the reef, and in identifying priority sites for removals.

Opportunistic lionfish monitoring also has considerable value as a means to quickly identify changes in the spatial distribution of lionfish, enabling the early detection of range expansions. Elsewhere in the Caribbean, recreational SCUBA divers have detected the introduction of alien marine species more quickly than fishers (Green et al. 2012a), while in North America citizen scientists have detected the range expansions of introduced crab species by monitoring coastlines on a far wider spatial scale than would have been possible by professional scientists (Darwall and Dulvy 1996; Branchini et al. 2015).

The opportunistic monitoring results do however suggest that there may have been an element of observer bias in the sighting of lionfish on different types of dive (Figure 2). The comparatively high number of lionfish sightings recorded on removal dives may be because culls were often conducted at sites known to have higher densities of lionfish, or it may be because volunteers were more engaged with or excited by lionfish searches on those dives and spread out over larger areas during the dive as lionfish detection and removal was the primary goal.

In contrast, LFS used a recognised and statistically rigorous method to collect accurate information about the density and biomass of lionfish. The time taken to conduct LFS was far greater than opportunistic sightings, reducing the potential time available for volunteers to conduct lionfish removals, or other coral reef monitoring activities (Darwall and Dulvy 1996). LFS also required a longer volunteer training period before data collection could commence. Lengthy training periods can deter citizen scientists from participating in projects (Sharpe and Conrad 2006) and repeating the same survey methods at similar sites can reduce volunteer interest and subsequently the quality of data (Goffredo et al. 2010). This is an important consideration as conservation volunteers typically travel at their own expense to assist with (and contribute to funding) survey work in return for fulfilment and knowledge (Goffredo et al. 2010; Silvertown et al. 2013). Protocols that are perceived to be strict or regimented may quickly reduce the appeal of a volunteering project, relative to alternative volunteering projects on offer, potentially reducing participants and funding (Carballo-Cárdenas and Tobi 2016).

Understanding the motivations of volunteers, and how those motivations may change throughout the duration of a project is therefore vital to ensuring those volunteers remain fulfilled, and that quality of data they collect remains high (Silvertown et al. 2013; Bird et al. 2014). For example, many volunteer SCUBA divers stopped contributing data to a lionfish monitoring initiative in the Dutch Caribbean after a year of participation as their perceptions about the recreational and/or commercial value of lionfish changed over that timeframe, and their initial motivations waned (Ward-Paige and Lotze 2011).

Despite the clear benefits that volunteers provide, the collection of ecological monitoring data by conservation volunteers presents some challenges. In accord with other citizen science projects (Silvertown et al. 2013), inter-observer variability was apparent and likely due to volunteers having differing levels of field and diving experience. More experienced divers with better buoyancy control are able to monitor and cull lionfish at a range of depths, under overhangs, on steep drop offs and in caves. For this reason, Blue Ventures has a prerequisite that conservation volunteers participating in lionfish expeditions already have a minimum of PADI Advanced Open Water certification or equivalent. However, the addition of more advanced diving experience as a prerequisite for conservation volunteers would reduce the potential pool of surveyors, and affect the financial sustainability of the programme. Some marine citizen science programs have not detected a significant difference in the results collected by experienced vs. inexperienced divers (Bird et al. 2014) with some even arguing that less experienced volunteers are better at following instructions and that the provision of training is the only important factor affecting volunteer performance (Butcher and Smith 2010). However the inclusion of quality assurance methods (e.g. a diver level scoring system to account to for inter-observer variability) will be explored as a priority in future (Popham 2015).

Based on our learnings from the LFS pilot year and subsequent power analysis, we have modified this LFS method by increasing the size of the search area to 50 m × 10 m, reduced replication at the site level, and increased the number of sites surveyed. Twenty-one permanent survey sites have been selected for annual surveys in BCMR, using a random number generator and a numbered grid laid over a map of BCMR.

The volunteer tourism sector is rapidly expanding, mainly in the form of gap year students (those taking a year off between school and university, or between university and starting a career) working on placements linked to conservation and community well-being (Credite Suisse et al. 2014). It has been estimated that 10 million volunteers a year are spending up to $2 billion on the opportunity to travel with a purpose (Popham 2015), providing an important additional source of income and manpower for conservation projects around the world. At a time when the funding available for conservation activities around the world is thought to be only 3–5% of what is required (Credite Suisse et al. 2014), and that the early detection of IAS offers the best chance of control (Lodge et al. 2007) our study suggests that conservation volunteers are a highly valuable resource to support marine IAS management activities.

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