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An assessment of two visual survey methods for documenting fish community structure on artificial platform reefs in the Gulf of Mexico

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ABSTRACT

Non-extractive visual survey methods are commonly used to assess a variety of marine habitats. The use of Underwater Visual Census (UVC) by SCUBA divers is predominant; however, remotely acquired video data (e.g., cameras systems, remotely operated vehicles (ROVs), submersibles) are becoming more frequently used to acquire community data. Both remote and diver-based surveys are currently used to survey artificial reef habitat in the Northwestern Gulf of Mexico (GOM) and have associated error due to inherent method bias. Because survey methods that most accurately document the occurrence and estimated abundance of several important fisheries species are greatly needed in the GOM, we compared data collected on the same days and sites from both Roving Diver Surveys (RDS) and micro-ROV surveys conducted on reefed oil and gas platforms. The combined datasets identified a total of 56 species from 22 families, and there was no significant difference in measured species richness between a comprehensive 30 min ROV survey and RDS. Five species of federally managed fish in the GOM were more frequently detected by ROV, as were the majority of species in the Lutjanid and Carangid families. However, abundance estimates from RDS surveys were up to an order of magnitude greater. Multivariate analyses indicated that method choice affected community composition, with Lutjanids and Carangids driving the differences. These two fish families in particular are subject to method bias, probably due to inflated abundance estimates with RDS, or alternatively, deflated estimates from ROV. Although our ROV surveys more frequently detected important fisheries species and produced conservative abundance estimates, a further examination of species distributions on these high-relief platform reefs is needed to fully determine the most accurate survey method. In addition, the attraction and/or gear avoidance of certain species to underwater vehicles deserves further investigation. Overall, our data indicate these methods are viable but the choice of survey method can have implications for the management of certain species, and that careful consideration of methodology is necessary to most accurately document species of interest.

1. Introduction

Researchers undertaking analyses of fish habitats often use data derived from visual survey methods to document the associated community in a non-destructive manner. Traditionally, these have included SCUBA diver based underwater visual census (UVC) methods. More recently, the increasing use of remote cameras systems, remotely operated vehicles (ROVs), and manned submersibles has given scientists access to areas out of the range of safe diving practices. However, the chosen survey technique may affect results due to method-specific consequences (i.e., bias) Differences between diver-based UVC methods have been commonly reported in the literature (Fowler, 1987; Bortone et al., 1989; Schmitt et al., 2002; Colvocoresses and Acosta, 2007; Consoli et al., 2007; Murphy and Jenkins, 2010; Dickens et al., 2011; Holt et al., 2013; Lindfield et al., 2014), and study-specific objectives will ultimately influence the chosen method. For instance, if there are particular species of interest, their behavior and reactions to divers may need to be evaluated before determining appropriate methods (Fowler, 1987; Ward-Paige et al., 2010; Bozec et al., 2011; Dickens et al., 2011). Certain methods may best document species richness, while others provide greater confidence with abundance and size estimation (Bortone et al., 1989; Schmitt et al., 2002; Guidetti et al., 2005; Tessier et al., 2005; Colvocoresses and Acosta 2007; Consoli et al., 2007; Bozec et al., 2011). Diver experience can also influence results (Williams

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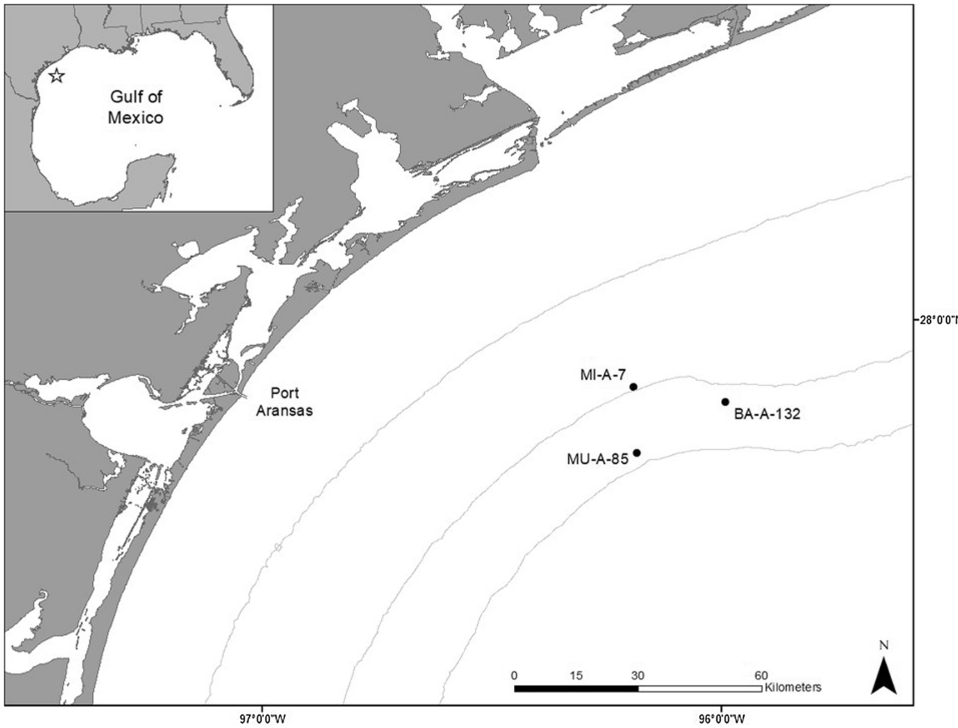

Fig. 1. Map of sampling region off South Texas depicting artificial reef (reefed oil and gas platforms) sampling sites identified by black dots. Bathymetry of region is displayed in increments of 30 m.

et al., 2006). Often, complimentary techniques are suggested to reduce sampling bias and gain insight into specific habitats (Schmitt et al., 2002; Consoli et al., 2007; Murphy and Jenkins, 2010; Bozec et al., 2011; Mallet et al., 2014).

Methods that minimize diver bias are also not without complication. Remotely deployed video-based devices have to be evaluated critically to determine what is best for a particular site or circumstance. Although these devices can be deployed in deeper habitats without time limitation, technical constraints (field of view, water clarity, use of bait, associated noise) can lead to counting errors and inhibit the interpretation of data. Baited video methods may influence the distance fish travel across habitats and even the species and numbers visible in the video frame (Willis et al., 2000; Cappo et al., 2003; Stobart et al., 2007; Harvey et al., 2007; Whitmarsh et al., 2018). Some species are known to be attracted or repelled by vehicle activity (reviewed in Stoner et al., 2008), which can cause error in estimates of abundance. Loss of field of view and a limited range of motion for ROVs may also result in lower detection of benthic or cryptic species (Cappo et al., 2006; Andaloro et al., 2013; Pita et al., 2014; Ajemian et al., 2015a). Although the use of video-based surveys removes the need for trained observers in the field, bottlenecks in laboratory data analysis can be problematic (Murphy and Jenkins, 2010). Additionally, the initial purchase costs of instrumentation and subsequent maintenance may be limiting. Ultimately, researchers face both logistical and financial restraints that create trade-offs in terms of choosing the most appropriate sampling methodology.

For artificial habitats in the Gulf of Mexico (GOM), scientists have used a variety of UVC and remote video-based methods to describe fish assemblages (Bortone et al., 1994; Rooker et al., 1997; Stanley and Wilson, 1997; Strelcheck et al., 2005; Lingo and Szedlmayer, 2006; Patterson et al., 2009; Dance et al., 2011; Ajemian et al. 2015a,b). However, as methodologies have been quite variable, opportunities to compare data sets between similar habitat types is are limited. In the Northwestern GOM, artificial reefs are often comprised of decommissioned oil and gas platforms, and Texas currently manages 210 platform reefs through the Texas Parks and Wildlife Department's Artificial Reef Program (TPWD-ARP). Since 1993, the fish community on many of these platforms has been monitored with SCUBA-based roving diver surveys (RDS) (Schmitt and Sullivan, 1996; Schmitt et al., 2002) which evaluate species composition and categorical abundance on these deep (≥ 30 m) structures. The RDS was originally developed to allow volunteer divers to quickly evaluate and record species seen on recreational dives (Schmitt and Sullivan, 1996). When used by highly trained divers, comparisons between RDS and other UVC methods have demonstrated the ability of RDS to document a greater number of species, and it has been recommended as complimentary to more traditional methods that document fish size and density (Schmitt et al., 2002; Holt et al., 2013).

In 2012, we began monitoring a group of platform reefs in the western GOM. Because of the large size and depths of these structures, specific sampling methodologies were developed to survey these sites using a micro-ROV (Ajemian et al., 2015a) in conjunction with SCUBA-based roving diver surveys. As a long-term RDS monitoring program exists for TPWD platform reefs, it is important to understand and evaluate sources of bias between these two methods, as well as how that bias may affect documentation of particular species. Direct comparisons of RDS and ROV-based surveys have not been undertaken to date; although, a few authors have compared other UVC methods to ROV at various locations. These studies suggest that ROVs may underestimate the number of species and abundances compared with UVC surveys (Carpenter and Schull, 2011; Andaloro et al., 2013; Pita et al., 2014) and indicate that researchers may need to make concessions for different life history strategies/reactions to ROVs if interested in certain species. In this study, we analyzed data from both RDS and micro-ROV surveys completed on the same days and at the same sites. We compared differences in species occurrence, estimates of abundance, and overall reef fish communities indicated by RDS and two types of ROV surveys (a short, surface-focused roving survey intended to replicate a RDS and a longer, more comprehensive survey than included both roving and depth interval sampling). Additionally, the survey/video processing effort and financial investment required for each method were evaluated in terms of sampling "cost" and effort. These results

Table 1
Reef sites and dates surveyed. Number of Roving Diver Surveys (No. RDS Surveys), Maximum depth of RDS surveys (Max RDS Depth), ROV (Remotely Operated Vehicle) survey time (ROV Struc. Time), maximum ROV survey depth (Max ROV Depth), water depth (Bottom Depth), reef orientation (Structure Type), and vertical relief of reef (Struc. Height) are included.

Site	Survey Date	No. RDS Surveys	Max RDS Depth (m)	ROV Struc. Time (min)	Max ROV Depth (m)	Bottom Depth (m)	Struc. Type	Struc. Height (m)
BA-A-132	9/25/13	6	31	24	44	61	topple	29
	10/8/13	6	31	26	57	61	topple	29
	8/5/14	6	30	29	61	61	topple	29
	9/8/14	5	32	41	58	61	topple	29
MI-A-7	8/10/13	5	30	26	56	60	cutoff	34
	10/8/13	6	30	32	58	60	cutoff	34
	8/5/14	6	32	31	58	60	cutoff	34
	9/8/14	7	31	25	39	60	cutoff	34
MU-A-85	9/25/13	6	32	30	74	84	cutoff	55
	8/5/14	5	30	26	50	84	cutoff	55
	9/8/14	7	35	31	62	84	cutoff	55

provide key information on the benefits and limitations of survey methods used to evaluate fish communities on similar habitat types. This comparison can enable researchers to select the most efficient and accurate sampling techniques for both structure type and species of interest.

2. Materials and Methods

2.1. Site descriptions and location

Three platform reef sites off the coast of Port Aransas, TX, were monitored in late summer/fall of 2013 and again in the fall of 2014 for a total of eleven sampling events (Fig. 1, Table 1). Sites were chosen to minimize variability in physical parameters with similar distance from shore and bottom depth. Vertical relief (top of structure to benthos) was also similar among structures. Although sites varied in structure orientation (partially removed versus toppled platforms), previous research (Ajemian et al., 2015b) has shown little difference in fish communities inhabiting these two structure orientations. These structures also fell within a bottom depth cluster (60 − 84 m) that was previously determined to share similar community composition (Ajemian et al., 2015b).

2.2. Equipment

A VideoRay Pro 4 micro-ROV equipped with a compass, depth sensor, temperature sensor, auto-depth holding capabilities, forward facing color camera (520 line, 0.1 lx), LED array for illumination, Lynn Photo enhancer software to enhance video in poor visibility, and laser scaler (8 cm between lasers) was used for surveys. The ROV was piloted with an integrated control box connected via a tether. Surface real-time observations were conducted with live feed from the camera (160° tilt and a 105° viewing angle). Depth and heading were visible on the real-time image screen. Because the VideoRay Pro 4 system did not record high-definition footage, we also mounted a GoPro© camera (HD Hero2) to the ROV for improved fish identification. The HD Hero2 filmed at 960p (30 fps) and had a 170° field of view. However, because GoPro cameras had restricted use and battery life, footage from these devices was used to solely supplement identification, with all counts conducted within the VideoRay field of view. The LED array was not used during surveys to eliminate possible bias due to light attraction (Stoner et al., 2008).

2.3. Survey methodology

2.3.1. Roving diver surveys

Divers trained in fish identification descended to the top of each structure (approximately 30 m depth) and after acclimating for a few minutes, performed two separate 5-minute RDS noting all species of fish they saw within their allotted time. In addition, divers noted the categorical abundance of each species throughout the dive, recording the final category for each species once they completed their dive on standardized survey sheets. These categories are essentially a logarithmic scale and include; Single (S), Few (F) – 2–10 fish, Many (M) – 11–100 fish, and Abundant (A) – greater than 100 fish. No limitations were placed on the divers as far as area to survey; however, divers were instructed to focus on larger bodied, more conspicuous species versus small cryptic and more sessile species (Blennies, Gobies, etc.). Data from survey sheets were entered into an electronic database within 48 h of completed dives. Generally, for each site and date surveyed, 5–7 surveys were completed for a total of 65 individual surveys. In addition to retaining all individual survey data, an aggregate density score (DEN) (Schmitt and Sullivan, 1996) was calculated for each species for each sampling event (by site/date; n = 11). This pooled data were then used for comparisons with ROV generated datasets from each event. The formula for DEN score is:

$$\mathrm{Den} = (\mathrm{S\ X\ 1}) + (\mathrm{F\ X\ 2}) + (\mathrm{M\ X\ 3}) + (\mathrm{A\ X\ 4})\ /\ n$$

where S, F, M and A correspond to species specific abundance categories each diver records on survey and n is the total number of diver surveys for each sampling event. The resulting value ranges from 1 to 4 and is generally representative of the mean abundance category for each species. The percentage of dives in which a species occurred (Sighting Frequencies; %SF) was also calculated for each species over all surveys (n = 65). Species were identified as rare (R; < 20 %), common (C; 20–70 %), or frequent (F; > 70 %) according to %SF (Schmitt and Sullivan, 1996). To account for the influence of survey order on species presence/absence, the order of Dive and ROV surveys were rotated within a sampling day with a minimum interval of 30 min between survey types.

2.3.2. ROV surveys

A detailed description of our methodology for surveying these large structures is presented in a previous study by Ajemian et al. (2015a). Briefly, upon deployment, the pilot immediately descended to the top of the artificial reef structure where a five-minute roving horizontal survey (ROV 5-min)—intended to mimic a diver-based RDS—was completed. The ROV then completed depth interval sampling towards the bottom of the reef along the down-current side of the structure. During this depth interval sampling, the ROV was held stationary with camera in the up-current direction for approximately 1 min at each 10 m increment. Upon reaching the maximum survey depth (determined by ambient depth or visibility limitations), the ROV was again piloted to span the outer surface area of the down-current side of the structure. It was necessary to avoid entering the interior and up-current side of reefs when strong currents were present to reduce the risk of entanglement. Data collected during the entire survey (ROV 5-min and the depth intervals) are considered the ROV-Total survey. Mean survey time was approximately 29 min. Eleven ROV 5-min surveys and 11 ROV-total surveys were completed.

2.4. Video analysis

Videos from the ROV recording systems (ROV standard, GoPro HD) were downloaded and analyzed with open-source video software (VLC™ media player) in the laboratory. Fish were identified to the lowest possible taxon, enumerated and recorded onto a spreadsheet each time they entered the field of view. Time of day, depth of occurrence, temperature and heading of ROV, and the time in and out of the water were recorded. A MaxN was generated for each species, which is the greatest number of individuals captured at any one time on the video. This conservative count represented the total number, at minimum, of individuals for a particular species during the survey and is the commonly preferred abundance metric reported for video survey data (Ellis and DeMartini, 1995; Willis et al., 2000; Watson et al., 2005; Merritt et al., 2011; Ajemian et al., 2015a). The MaxN was converted to categorical data to facilitate comparisons with diver generated data. Categories were then converted into numeric representations (S = 1, F = 2, M = 3 and A = 4), essentially a log-scale transformation. Sighting frequency (%SF) was also generated (as described above) from ROV data. Data from the ROV 5-min surveys and the ROV-total surveys were compared with the diver generated RDS data. Ancillary data from additional ROV surveys at these sites was used to evaluate common depth distributions of five species of commercial and recreational importance (Gray Triggerfish, Gray Snapper, Vermilion Snapper, Greater Amberjack, and Red Snapper).

2.5. Univariate analysis

Number of species identified by each method (RDS, ROV-total, and ROV-5 min) were compared for each sampling event (n = 11) and overall. To determine if order of survey (first or second) influenced the number of species identified, mean values were calculated for the following categories: individual dive surveys (RDS-Ind; n = 65), aggregate dive surveys for each event (RDS; n = 11), total ROV survey for each event (ROV-total; n = 11), and five minute ROV surveys for each event (ROV-5 min; n = 11). An overall mean species richness for sampling order (first or second) was also calculated. Differences in the mean number of species identified by each method/order were evaluated by independent t-tests or analyses of variance (ANOVA) in SigmaPlot 13.0. Data were tested for normality and homogeneity of variance prior to statistical analysis. The Bonferroni method was used for pairwise comparisons as necessary. To determine if survey method influenced reported abundance indices, differences for species noted on the same day and site for at least three sampling events were compared. Converted minimum species counts from ROV-total surveys were compared to the RDS aggregate DEN score for species that met the criteria.

2.6. Multivariate analysis

Community assemblages identified for each sampling event and method (n = 33) were compared using PRIMER *v7* (Clarke et al., 2014a). The species specific MaxN data generated from ROV-total and ROV-5 min surveys were converted into categorical data for comparison to aggregate DEN scores generated from SCUBA sampling. Species accumulation curves were created using a Michaelis-Menton (MM) model for each method to determine if the number of surveys was adequate to describe the fish community present. Prior to generating a Bray-Curtis resemblance matrix to compare assemblages across samples, no transformation of data was done as categorical data conversions and DEN scores essentially result in log-transformed data. A two-dimensional non-metric Multi-Dimensional Scaling (*n*MDS) plot was created from the Bray-Curtis similarity matrix for all events and methods (n = 33). After review of this 2D ordination, a two-way permutational MANOVA (PERMANOVA; Anderson, 2001) was used to assess community differences with both site and method as factors and to test for an interaction between those factors. Because the interaction effect between method and site was not statistically significant (effects of each factor are not dependent on the other), method was further examined individually. Pairwise tests between methods were completed to determine groups responsible for assemblage variability. A similarity percentages routine (SIMPER; Clark 1993) was used to analyze species contributing the most to the dissimilarity between methods. The resulting community assemblage was then compared through a *n*MDS means plot using a Bray Curtis similarity matrix averaging across site for each method (n = 9). Hierarchical Cluster analysis was also used to further evaluate similarity between groups. Patterns of averaged relative abundance were also displayed visually across method with the use of a shade plot which presents the data in a linearly increasing gray scale. The use of this visual representation can help determine which species may be influencing multivariate results (Clarke et al., 2014b). Only species that contributed greater than 5 % to the abundance in each sample were included in this analysis.

3. Results

Total sampling effort was approximately equal between RDS and ROV-Total surveys (325 and 321 min, respectively), although the total number of individual surveys required was much greater for RDS (n = 65 vs. n = 11). Effort for ROV-5 min surveys was substantially lower (55 min). A total of 56 species representing 22 families were identified after summing all methods and sampling events (Table 2). Overall, RDS identified the greatest number of species (49), with ROV-5-min identifying the least (36) species. ROV-Total was comparable to RDS with 46 species identified. A randomly permutated species accumulation plot for samples combined (RDS + ROV-Total) showed a curve approaching an asymptote (Michaelis-Menton $S_{\max} = 60.94$) (Fig. 2), indicating that the combination of methods was able to most accurately describe the fish community at these sites. Curves generated from the three individual methods indicated that RDS and ROV-total similarly required more sampling effort to adequately describe the community (Michaelis-Menton $S_{\max} = 53.6$ and 51.5, respectively). However, the ROV-5 min data curve fell well below the other methods (Michaelis-Menton $S_{max} = 42.52$). The ROV-5 min survey consistently identified fewer species when compared to pooled RDS survey data for sampling events, and the mean species richness (12) was significantly lower for that method when compared to the others (One-way ANOVA, $F_{2,30} = 15.013$, $p < 0.001$). However, a comparison of these short ROV surveys to single diver surveys indicated both methods identified equal numbers of species (t-test; $t(74) = 0.565$, $p = 0.574$). The method identifying the greatest number of species for each event varied between RDS and ROV-Total, and mean species richness for these two methods (21 and 20, respectively) was not different (One-Way ANOVA, $F_{2,30} = 15.013 =$, $p < 0.001$). An analysis of proportion of structure the ROV-total survey covered indicated that higher species counts (> 20 species identified) were more common when greater than 80 % of the structure relief was surveyed (Fig. 3). Although survey methods performed first typically had a slightly higher species count (Fig. 4), the mean species richness values for all survey types grouped by order ($\bar{x} = 21.54 \pm 3.93$ and 19.09 ± 2.70) were not significantly different (t-test; $t(20) = 1.706$, $p = 0.103$).

The greatest number of unique species (10, Table 2) was documented through RDS generated data, and included Bar Jack (*Caranx ruber*), Bicolor Damselfish (*Stegastes paritus*), Doctorfish (*Acanthus chirurgus*), Gray Angelfish (*Pomacanthus arcuatus*), Graysby (*Cephalopholis cruentatus*), Lane Snapper (*Lutjanus synagris*), Scalloped Hammerhead (*Sphyrna lewini*), Spanish Mackerel (*Scomberomorus maculatus*), Yellowtail Snapper (*Ocyurus chrysurus*) and Warsaw Grouper (*Hyporthodus nigritus*). Comparatively, ROV methods identified seven

Table 2
Sighting Frequency (SF%), the percentage of dives in which a species occurred, for all species identified for each method. Rare (R, < 20 %), Common (C; 20–70 %) or Frequent (F; > 70 %) species are identified according to method.

Family	Common Name	Scientific Name	RDS		ROV - 5min		ROV - total	
Acanthuridae	Blue Tang	*Acanthurus coeruleus*	37	(C)	64	(C)	64	(C)
	Doctorfish	*Acanthurus chirurgus*	2	(R)	–		–	
Balistidae	Gray Triggerfish	*Balistes capriscus*	6	(R)	9	(R)	18	(R)
Carangidae	African Pompano	*Alectis ciliaris*	6	(R)	–		27	(C)
	Almaco Jack	*Seriola rivoliana*	25	(C)	–		27	(C)
	Bar Jack	*Caranx ruber*	5	(R)	–		–	
	Black Jack	*Caranx lugubris*	2	(R)	9	(R)	27	(C)
	Blue Runner	*Caranx crysos*	34	(C)	27	(C)	45	(C)
	Crevalle Jack	*Caranx hippos*	45	(C)	27	(C)	64	(C)
	Greater Amberjack	*Seriola dumerili*	51	(C)	45	(C)	82	(F)
	Horse-eye Jack	*Caranx latus*	14	(R)	27	(C)	64	(C)
	Lookdown	*Selene vomer*	2	(R)	–		18	(R)
	Permit	*Trachinotus falcatus*	3	(R)	9	(R)	9	(R)
	Rainbow Runner	*Elagatis bipinnulata*	34	(C)	27	(C)	55	(C)
	Yellow Jack	*Caranx bartholomaei*	9	(R)	9	(R)	55	(C)
Carcharhinidae	Sandbar Shark	*Carcharhinus plumbeus*	–		9	(R)	45	(C)
Chaetodontidae	Reef Butterflyfish	*Chaetodon sedentarius*	2	(R)	18	(R)	45	(C)
	Spotfin Butterflyfish	*Chaetodon ocellatus*	20	(C)	27	(C)	36	(C)
Ephippidae	Atlantic Spadefish	*Chaetodipterus faber*	20	(C)	27	(C)	27	(C)
Epinephelidae	Atlantic Creolefish	*Paranthias furcifer*	91	(F)	100	(F)	100	(F)
	Black Grouper	*Mycteroperca bonaci*	–		–		9	(R)
	Graysby	*Cephalopholis cruentatus*	9	(R)	–		–	
	Rock Hind	*Epinephelus adscensionis*	80	(F)	27	(C)	82	(F)
	Scamp	*Mycteroperca phenax*	3	(R)	–		18	(R)
	Warsaw Grouper	*Hyporthodus nigritus*	2	(R)	–		–	
Haemulidae	Tomtate	*Haemulon aurolineatum*	–		–		9	(R)
Holocentridae	Squirrelfish	*Holocentrus adscensionis*	–		–		9	(R)
Kyphosidae	Bermuda Chub	*Kyphosus sectatrix*	45	(C)	64	(C)	64	(C)
Labridae	Bluehead	*Thalassoma bifasciatum*	75	(F)	73	(F)	82	(F)
	Creole Wrasse	*Clepticus parrae*	8	(R)	9	(R)	9	(R)
	Spanish Hogfish	*Bodianus rufus*	92	(F)	100	(F)	100	(F)
	Spotfin Hogfish	*Bodianus pulchellus*	85	(F)	100	(F)	100	(F)
Lutjanidae	Cubera Snapper	*Lutjanus cyanopterus*	2	(R)	18	(R)	36	(C)
	Gray Snapper	*Lutjanus griseus*	74	(F)	73	(F)	100	(F)
	Lane Snapper	*Lutjanus synagris*	2	(R)	–		–	
	Red Snapper	*Lutjanus campechanus*	42	(C)	18	(R)	82	(F)
	Vermilion Snapper	*Rhomboplites aurorubens*	37	(C)	45	(C)	82	(F)
	Yellowtail Snapper	*Ocyurus chrysurus*	2	(R)	–		–	
Pomacanthidae	Blue Angelfish	*Holacanthus bermudensis*	86	(F)	73	(F)	100	(F)
	French Angelfish	*Pomacanthus paru*	11	(R)	9	(R)	27	(C)
	Gray Angelfish	*Pomacanthus arcuatus*	3	(R)	–		–	
	Queen Angelfish	*Holacanthus ciliaris*	23	(C)	9	(R)	18	(R)
	Rock Beauty	*Holacanthus tricolor*	2	(R)	9	(R)	9	(R)
	Townsend Angelfish	*Holacanthus sp.*	2	(R)	9	(R)	18	(R)
Pomacentridae	Bicolor Damselfish	*Stegastes partitus*	3	(R)	–		–	
	Brown Chromis	*Chromis multilineata*	–		9	(R)	9	(R)
	Sergeant Major	*Abudefduf saxatilis*	38	(C)	36	(C)	55	(C)
Priacanthidae	Bigeye	*Priacanthus arenatus*	–		–		9	(R)
Rachycentridae	Cobia	*Rachycentron canadum*	3	(R)	9	(R)	18	(R)
Scombridae	Atlantic Bonito	*Sarda sarda*	5	(R)	–		9	(R)
	Spanish Mackerel	*Scomberomorus maculatus*	2	(R)	–		–	
Scorpaenidae	Red Lionfish	*Pterois volitans*	5	(R)	9	(R)	27	(C)
Serranidae	Whitespotted Soapfish	*Rypticus maculatus*	–		9	(R)	9	(R)
Sphyraenidae	Great Barracuda	*Sphyraena barracuda*	88	(F)	82	(F)	100	(F)
Sphyrnidae	Scalloped Hammerhead	*Sphyrna lewini*	2	(R)	–		–	
Tetraodontidae	Sharpnose Puffer	*Canthigaster rostrata*	15	(R)	–		18	(R)

unique species; Bigeye (*Priacanthus arenatus*), Black Grouper (*Mycteroperca bonaci*), Squirrelfish (*Holocentrus adscensionis*), Tomtate (*Haemulon aurolineatum*), Whitespotted Soapfish (*Rypticus maculatus*), Sandbar Shark (*Carcharhinus plumbeus*), and Brown Chromis (*Chromis multilineata*). All ten species unique to diver-based RDS surveys were noted as rare; with less than 20 % sighting frequency. Unique species identified through ROV methods were all also identified as rare with the exception of Sandbar Shark which was commonly sighted in ROV-total surveys (45 % SF).

Species considered to be frequent (F) in occurrence on reef sites across all methods included Atlantic Creolefish (*Paranthias furcifer*), Blue Angelfish (*Holacanthus bermudensis*), Bluehead (*Thalassoma bifasciatum*), Gray Snapper (*Lutjanus griseus*), Great Barracuda (*Sphyraena barracuda*), Spanish Hogfish (*Bodianus rufus*)and Spotfin Hogfish (*Bodianus pulchellus*) (Table 2). Eight other species were considered common (C) in occurrence across all methods. In several instances, species were documented as common on ROV-total surveys and rare or absent with the other methods. These species include French Angelfish (*Pomacanthus paru*), Reef Butterflyfish (*Chaetodon sedentarius*), Red Lionfish (*Pterois volitans*), Black Jack (*Caranx lugubris*), Yellow Jack (*Caranx bartholomaei*), African Pompano (*Alectis ciliaris*), Cubera Snapper (*Lutjanus cyanopterus*), and Sandbar Shark (*Carcharhinus plumbeus*). Greater Amberjack (*Seriola dumerili*), Red Snapper (*Lutjanus campechanus*) and Vermilion Snapper(*Rhomboplites aurorubens*) were documented as frequent (> 70 %) with ROV-Total surveys; however, RDS and ROV-5 min surveys appear to under-report as they were noted

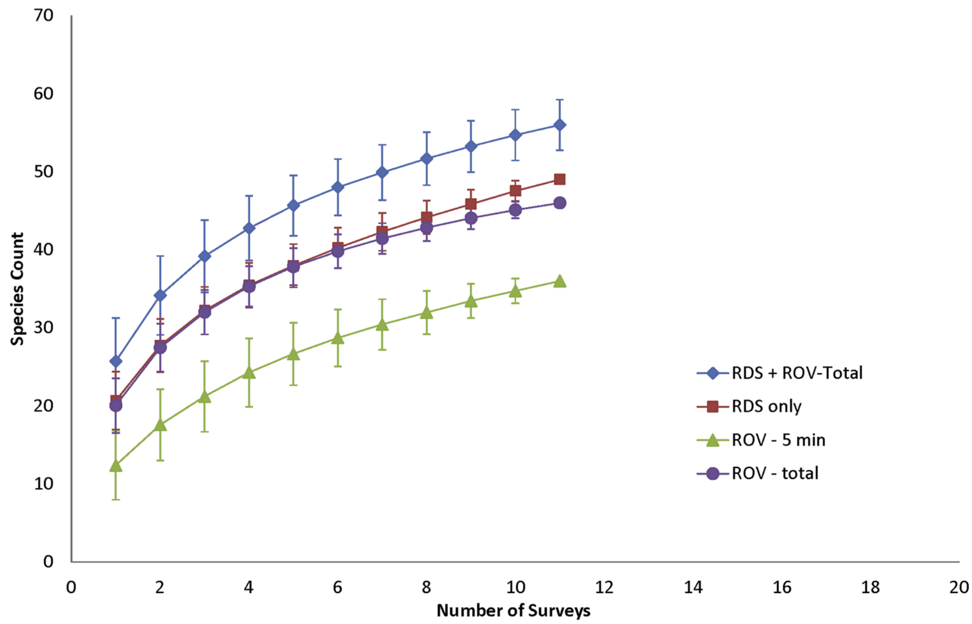

Fig. 2. Randomly permutated species accumulation plot with standard deviation for samples combined from RDS + ROV-total, RDS only, ROV-5 min, and ROV-total. Michaelis-Menton Smax values were 60.94, 53.6, 51.5, and 42.52 respectively.

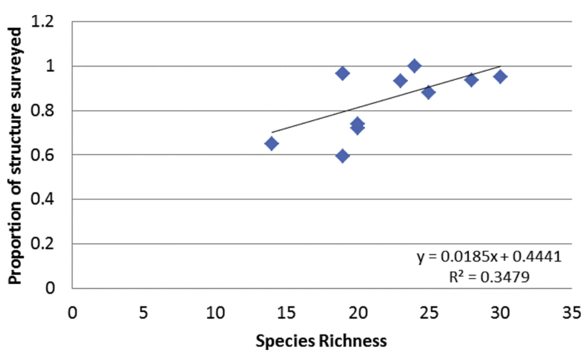

Fig. 3. Species counts for ROV-total surveys compared with proportion of structure surveyed.

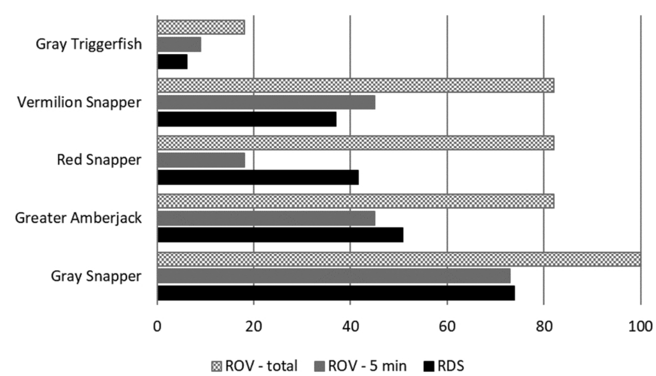

Fig. 5. Sighting Frequency (%) by method of five species of commercial and recreational importance.

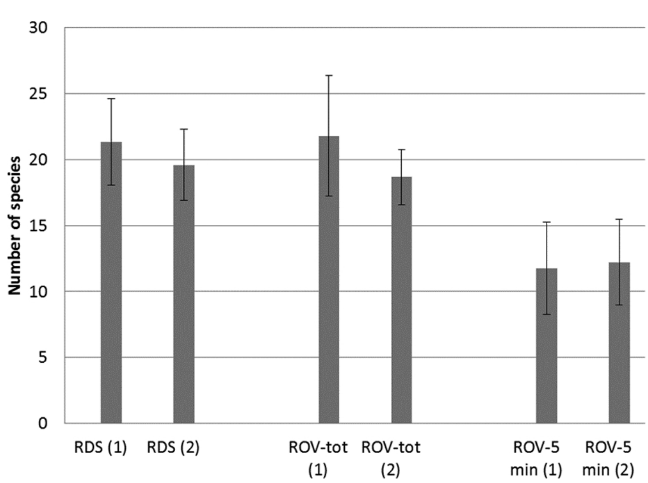

Fig. 4. Mean number (± SD) of species identified by method and survey order (1, first or 2, second). Means were computed for dive summary data for each sampling event (RDS; n = 11), total ROV surveys for each sampling event (ROV-total; n = 11), and 5-minute ROV surveys for each sampling event (ROV-5 min; n = 11).

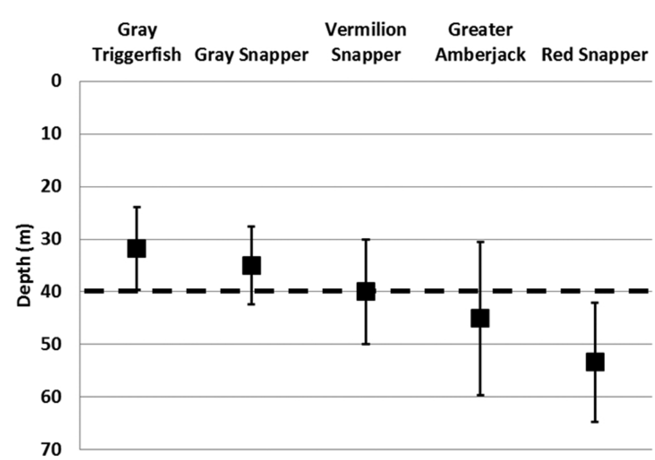

Fig. 6. Mean ± SD of depth distribution for five federally managed species. This data was derived from all ROV surveys completed in 2012–2014 at the three platforms reefs, MI-A-7, BA-A-132, and MU-A-85. The limit (40 m) for most non-technical scientific diving is noted with a dashed line.

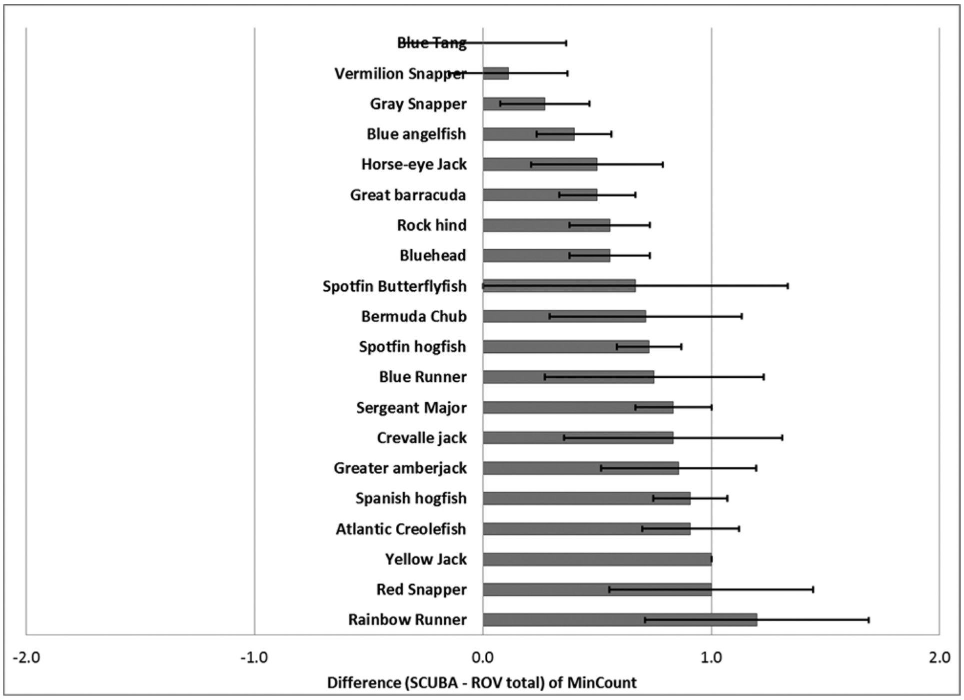

Fig. 7. Difference +-SE of mean abundance data generated thru RDS and ROV-total surveys. ROV-total converted MaxNs were subtracted from SCUBA DEN aggregate scores for each species when identified on same days and sites for at least 3 sampling events. Zero indicates equal abundances. Positive values indicate RDS abundance index was greater.

as common (20–70 %) or rare (< 20 %) (Fig. 5). These differences in detection may be an important consideration for surveys focused on these economically valuable GOM fisheries species. In fact, with the exception of two rarely sighted species (Yellowtail, *Ocyurus chrysurus* and Lane Snapper, *Lutjanus synagris*), RDS sighting frequencies of Lutjanids were always 30–40 % less than with ROV-total. Carangids followed a similar pattern, with the majority of species sighted much more frequently with ROV-total surveys (Table 2). An additional analysis of ancillary depth distribution data indicated that Greater Amberjack and Red Snapper were most commonly observed out of normal SCUBA range (> 40 m; Fig. 6).

For abundance comparisons, a total of 20 species were observed on the same days and sites for at least three sampling events. (Fig. 7). Of these, only one species (Blue Tang, *Acanthurus coeruleus*) had equal abundance estimates across methods. For the other 19 species, RDS estimates exceeded ROV-total up to one order of magnitude. Mean abundance estimates for five species of commercial/recreational importance (Fig. 8) indicate that ROV-5 min surveys are consistently lower than the other methods. Although, RDS and ROV-total are more comparable, abundance estimates from diver-based surveys are higher

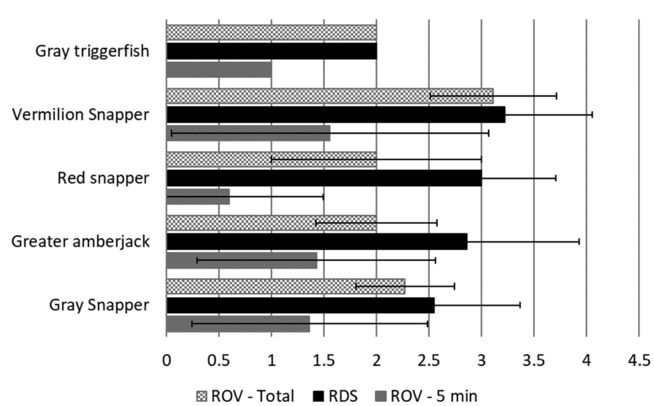

Fig. 8. Comparison of mean abundance indices ± SD from each method for species of commercial/recreational importance. These mean values were calculated when species occurred on same site and date for at least 3 sampling events.

for four of the five species.

An initial comparison of the community assemblage through an nMDS plot revealed possible community similarities explained by both method and site, although the stress was high (2D stress = 0.23), indicating difficulty in displaying relationships through this method. The subsequent two-way PERMANOVA revealed significant differences for both method ($F_{2,32}$ = 5.7321, P = 0.001) and site ($F_{2,32}$ = 6.5082, P = 0.001); however, there was no interaction between the two factors ($F_{4,\ 32}$ = 0.6401, P = 0.936). Resulting two-way PERMANOVA comparisons for method indicated that major differences were attributed to those between ROV-5 min surveys and RDS (t = 3.2305, P (perm) = 0.001). Results from the SIMPER analysis identified species belonging to the Carangid (Blue Runner, Rainbow Runner and Greater Amberjack) and Lutjanid (Red Snapper and Vermilion Snapper) families as those most responsible for community differences between methods, although none contributed more than 6.5 % to the observed differences (Table 3). Similar to previous abundance comparisons (see Fig. 8), RDS documented higher abundances of these species, with the exception of Red Snapper. When all sampling events were analyzed with SIMPER, overall Red Snapper abundance was greater with ROV-total surveys than RDS. Differences in method were more apparent in the resulting nMDS means plot (Fig. 9), with less stress, and indicate the closer relationship between RDS and ROV-Total surveys. A shade plot clearly presents the increase in documented abundance metrics apparent for RDS generated data (Fig. 10). Seven species previously noted to occur frequently on these reef sites across all methods (Atlantic Creolefish, Blue Angelfish, Bluehead, Gray Snapper, Great Barracuda, Spanish Hogfish, Spotfin Hogfish) are also apparent in this plot and are clustered together, indicating their common occurrence and similar habitat use. Horse-eye Jack, Greater Amberjack, and Blue Tang also occurred across all sites and methods; although they were less abundant than most other taxa identified. The other species noted as important (> 5 % of total) for any one sample included Blue Runner, Bermuda Chub, Rainbow Runner, Vermilion Snapper, Red Snapper and Atlantic Spadefish, although these species were absent from at least one or more survey type/site and as a result were responsible for most of the dissimilarity between methods. Specifically, the abundance metrics clearly show higher values for RDS for both Blue and Rainbow Runner, two species identified as most dissimilar for RDS and ROV methods. The

Table 3
Results of SIMPER analysis of community differences due to method. The top three species identified as contributing the most to dissimilarity between groups are listed along with percent contribution. Data from all sampling events was included (n = 33). Abundance indices are based on aggregate DEN scores (RDS) or converted MaxN data (ROV).

RDS vs. ROV-total Species	RDS Abund.	ROV-Tot. Abund.	Average Diss.	Diss. / SD	Contrib. %	Cumm. %
Blue Runner	2.06	1.09	2.08	1.22	4.83	4.83
Rainbow Runner	2.12	1.09	2.03	1.34	4.71	9.53
Red Snapper	1.48	1.73	1.78	1.37	4.12	13.66
RDS vs. ROV-5 min Species	**RDS Abund.**	**ROV- 5 m. Abund.**	**Average Diss.**	**Diss. / SD**	**Contrib. %**	**Cumm. %**
Rainbow Runner	2.12	0.45	3.00	1.42	5.55	5.55
Vermilion Snapper	2.55	1.27	2.77	1.35	5.13	10.68
Blue Runner	2.06	0.55	2.71	1.24	5.01	15.69
ROV-Total vs. ROV-5 min Species	**ROV-Tot. Abund.**	**ROV- 5 m. Abund.**	**Average Diss.**	**Diss. / SD**	**Contrib. %**	**Cumm. %**
Vermilion Snapper	2.55	1.27	3.18	1.20	6.52	6.52
Red Snapper	1.73	0.27	2.68	1.45	5.50	12.01
Greater Amberjack	1.73	0.91	2.35	1.22	4.81	16.82

higher abundance indices seen in the SIMPER analysis for Red Snapper measured with ROV-total appear to be due to a lack of detection with RDS at one site (MU-A-85). Additionally, Red Snapper and Vermilion Snapper were identified through SIMPER as driving dissimilarity between RDS, ROV-Total and ROV-5 min datasets, and both species were absent from ROV-5 min surveys at several sites. Vermilion Snapper in particular, was documented with much higher abundance indices for ROV-total and RDS than with ROV-5 min.

4. Discussion

The comparison presented here is the first to evaluate fish community visual survey data derived from roving diver surveys (RDS) and a micro-ROV on offshore platform reefs. A comparison of these specific methods is important, as renewed interest in generating detailed community and fish abundance data is important for a better understanding of the ecological role these structures play and their subsequent management. In this study, both methods documented a variety of species inhabiting platform reefs, although differences due to survey method were evident. Survey method appeared to influence frequency and abundance metrics as well as the fish community analysis.

A compilation of data from the RDS and ROV-based surveys was able to document a significant portion of the fish assemblage. The total number of species identified (56) was comparable to previously reported fish species richness (59) identified by ROV surveys on artificial reefs in the same area with a greater number of sites (n = 15) and surveys (n = 44) (Ajemian et al., 2015b), indicating the utility of combining RDS and ROV methodologies. Roving diver surveys have been noted as the preferred UVC method if species documentation is the study goal (Schmitt et al., 2002; Toller et al., 2010; Holt et al., 2013), and the use of an ROV is typically thought to result in less species identified when compared to UVC methods (Cappo et al., 2006; Andaloro et al., 2013; Pita et al., 2014; Ajemian et al., 2015a), especially with regard to benthic or cryptic species. It is well known that both size and behavior of species influence UVC derived reports and abundance estimates, and the effect would perhaps be greater with ROV due to the loss of a significant portion of the viewing field (Bozec et al., 2011). For effective comparisons of these two specific methods (RDS and ROV-5 min) the area of survey was constrained, and because survey time directly impacts number of species reported (Tessier et al., 2005) surveys were limited to five-minute intervals. Additionally, the divers in this study generally did not document small-bodied cryptic or benthic species (Gobies, Blennies, etc.) which may be difficult to identify on ROV video; instead larger, more conspicuous species dominated survey results. Nonetheless, as expected, surveys conducted by dive teams always outperformed the singular ROV-5 min survey.

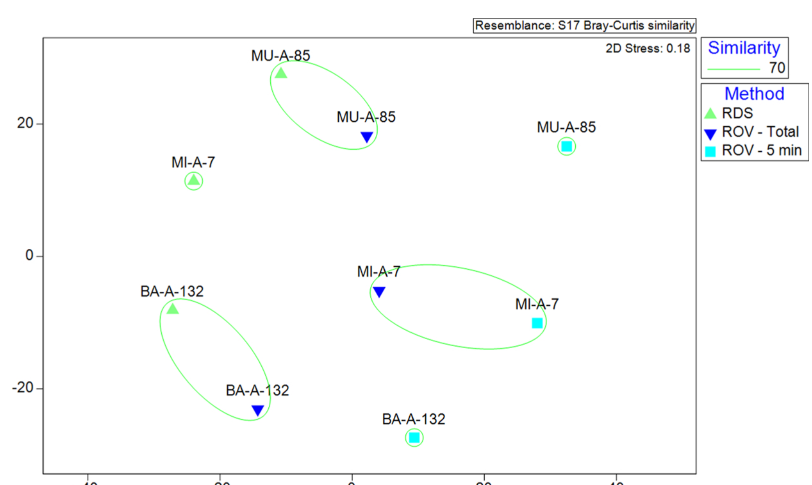

Fig. 9. Non-metric multidimensional scaling (nMDS) plot created from the Bray Curtis dissimilarity matrix of mean values for each site by method. Similarity between groups through Hierarchical Cluster analysis is also indicated by circles.

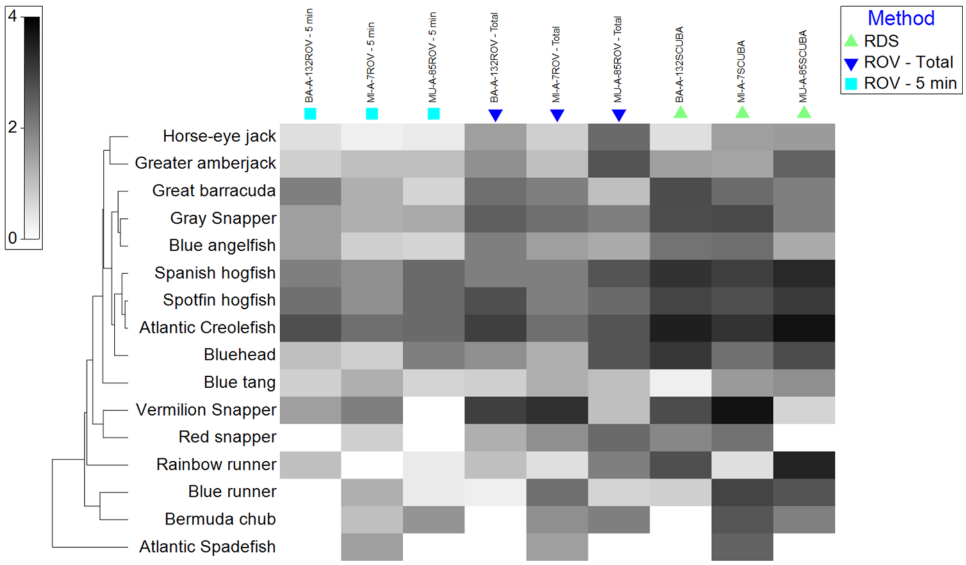

Fig. 10. Shade plot for DEN aggregate scores and converted MaxNs for species that accounted for at least 5 % of abundance in any one sample (n = 16). Data was averaged across sites by method. The species axis is ordered according to cluster analysis thru Whittaker's Index of Association (Clarke et al 2014).

This is not surprising as effort for dive teams was four to six times greater than for a single ROV-5 min survey. However, we also found that a ROV-5 min survey closely approximates a single diver roving diver survey, perhaps indicating similar performance when documenting conspicuous species. Even with such reduced effort, ROV-5 min surveys were still able to document approximately 64 % of the total species richness observed for all methods combined, suggesting these quick, roving surveys may have utility in some instances.

When comparing the longer, more comprehensive ROV-Total surveys to RDS, although there was no significant difference in measured species richness, each method documented a number of unique species. Several common schooling pelagic species (Bar Jack, Spanish Mackerel and Scalloped Hammerhead) were observed only by SCUBA, which may indicate a diver's larger field of view, as they can more easily document species that may remain farther away from the structure (Andaloro et al., 2013; Tessier et al., 2005). In fact, unpublished data from numerous ROV surveys in the same area over a longer time period (Wetz, unpublished data) have never documented Spanish Mackerel, Scalloped Hammerhead or Warsaw Grouper, perhaps indicating an avoidance to this technology by these species (Stoner et al., 2008), or alternatively an attraction to divers. Smaller-bodied species such as Bicolor Damselfish and Graysby tend to be more closely associated with the structure and often are under ledges or structure, which may interfere with direct observation by ROV. Divers did not note Black Grouper, Brown Chromis, Squirrelfish, Tomtate or Whitespotted Soapfish on surveys, but these species have been documented on previous SCUBA surveys in the same area (Wetz, unpublished data), probably indicating their rare occurrence not diver avoidance. Bigeyes were also only identified on the ROV video (60 – 61 m depth), but are typically classified as deeper dwelling species in the northwestern GOM (Dennis and Bright, 1988). Surprisingly, Sandbar Sharks, although commonly observed on our ROV surveys, were not documented on a SCUBA survey, perhaps indicating an attraction to the motion, noise or electrical impulses of the ROV (Stoner et al., 2008). Ancillary ROV data from this region (Wetz, unpublished) has documented the Sandbar Shark on multiple occasions, but typically at depths inaccessible to divers (40 – 70 m), perhaps also indicating a deeper depth preference on these structures. A previous analysis of ROV data in this area has shown that the entire vertical expanse of the structure should be surveyed to document the greatest species richness and showed higher species accumulation rates at the top and bottoms of these structures (Ajemian et al., 2015a). In the present study, although the ROV-Total surveys were able to document species over a greater depth range than RDS, due to the regional nepheloid layer (Shideler, 1981) and associated visibility constraints only 55 % of ROV surveys covered more than 90 % of the entire structure relief, with only one survey reaching the benthos. There was a positive correlation with species richness documented and proportion of structure surveyed, and additional surveys that spanned the full vertical relief may have increased the number of species detected by ROV.

The most interesting difference between RDS and ROV documentation is the discrepancy between frequency and abundance metrics, which may have management implications for certain species. Because of the variety of methods used to monitor fish populations throughout the GOM, comparisons of datasets and inclusion of those datasets into the management process is complicated and rarely done. The human eye is generally thought to be far superior when assessing fish underwater (Cappo et al., 2003; Murphy and Jenkins, 2010); however, a number of previous studies have demonstrated bias in both methods, depending on species-specific reactions to both divers and ROVs (Bozec et al. 2011; Carpenter and Shull, 2011; Dickens et al., 2011; Pita et al., 2014; Parrish and Boland, 2004; Willis et al., 2000; Stanley and Wilson, 1995). Our data show that method choice can have a profound impact on both abundance indices and frequency data even when sampling occurs on the same days/sites. Five species of federally managed fish in the GOM were much more frequently detected by ROV, as were the majority of species in the Lutjanid and Carangid families (Fig. 5). This is likely due to the average depth distributions of many of these species being predominantly below the safe diving limit (Fig. 6). In addition, perhaps these species are shy of divers as previous data from petroleum platforms in the GOM have shown that fish density decreased by 41–77 % when divers entered the water (Stanley and Wilson, 1995). Alternatively, our data could have been influenced by attraction to the ROV for some species. Gray Triggerfish (*Balistes capriscus*) often exhibit territorial behavior in response to ROV presence (Wetz, personal observation; Patterson et al., 2009; Simmons and Szedlmayer, 2012) and actively pursue the vehicle. Lutjanid species such as Greater Amberjack, Horse-eye Jack (*Caranx latus*), and Almaco Jack (*Seriola rivoliana*) have often been observed circling and following the ROV (Wetz, personal observation). Fish attraction to underwater vehicles is not uncommon and has been noted for approximately 43 % of the studies reviewed by Stoner et al. (2008), which included rockfishes, flatfishes, and hakes. This behavior could bias results and lead to overestimations of abundance in some instances; however, in this case, the use of the conservative MaxN metric should minimize overestimations (Cappo et al., 2003). Because of the importance of several species in the Lutjanid family to GOM fisheries, particularly Red Snapper, the discrepancy in detection is a valid concern for future analyses and one managers should consider. Our analysis of number of species identified versus order of survey method showed no differences, indicating that attraction/repellence may not be influencing results. However, depth does likely influence the differences noted as two of these five species are most commonly sighted at depths outside of the normal dive survey range (Fig. 6). For Red Snapper in particular, which are known for benthic-associated lifestyles (Gallaway et al., 2009), our data indicate that SCUBA may not be the proper monitoring choice for this species. Further demonstrating the importance of deep surveys on these large structures, invasive Lionfish (Pterois sp.) were documented almost six times as frequently on ROV-total versus RDS surveys. In fact, the earliest confirmed sighting in the Texas Coastal Bend Region was by ROV, at a depth inaccessible to typical SCUBA surveys (66 m; Ajemian et al., 2015b).

When comparing abundance estimates, RDS survey estimates were up to an order of magnitude greater for the majority of species compared during the same sampling event. The limited field of view of the ROV may explain this difference as divers have a much larger area from which they can enumerate fish present on the site. Additionally, divers may count the same school of fish several times during a survey, artificially inflating abundance for some species (Tessier et al., 2005). Previous authors comparing ROV and UVC methods also found that ROV surveys were less precise and underestimated less abundant, cryptic and nekto-benthic fish (Pita et al., 2014; Andaloro et al., 2013) while other species were overestimated by ROV. Although divers in the present study did not focus on small cryptic species, we did see increased abundance estimates for the majority of species with RDS. However, these roving diver surveys are not meant to accurately estimate fish density. For those who wish to accurately document this metric, the addition of another UVC method such as SPC or belt transects is likely more appropriate. Our data indicate that the ROV is perhaps underestimating abundance metrics specifically when compared to RDS; however, video is known to be a conservative measure with the use of the MaxN metric and can eliminate errors often due to double-counting (Ellis and DeMartini, 1995; Cappo et al., 2003).

Because sites were chosen to minimize variability in physical parameters, and a previous study at those same sites indicated similar community composition (Ajemian et al., 2015b), community differences were not expected. The PERMANOVA analysis in the present study indicated there was no interaction between method and site; therefore, we focused on the community differences identified through survey method and considered site differences to be outside the scope of this study. As with previous comparisons, species of Lutjanids and Carangids drove the differences between methods, indicating that these families in particular are subject to method bias, probably due to variance in abundance estimates. Results clearly identified a group of common artificial reef resident species that appear to have similar behaviors and habitat use allowing for consistent detection across sites and methods, although greater abundances for these species are still indicated for RDS methods. For the species driving major differences in communities due to survey method (Blue Runner, Rainbow Runner, Red Snapper, Vermilion Snapper), presence/absence and magnitude of abundance differences clearly can be attributed to the schooling behaviors and patchy distributions associated with these species (Ajemian et al., 2015a). Reactions of fish to divers and ROV may also influence measurements. For such species, a variety of sampling methods may increase accuracy in evaluating their populations.

A comparison of the initial financial investment required to initiate sampling with either method indicated that micro-ROV based methods may cost up to three times that of a diver based RDS method, depending on equipment choice. Although the base cost of a similarly outfitted ROV can vary widely, a unit such as the one described here (depth rating of 300 m + accessories) costs approximately $40,000 USD and is at the upward end of micro-ROV investment. Costs for dive equipment (tanks, gear, etc.) are estimated at $14,000 USD for six scientific divers. However, other factors can influence true project costs. A greater number of field personnel are required for diver-based sampling due to limitations on dive times, depths, and scientific diving guidelines. We routinely had six scientific divers plus a boat captain on all sampling trips, while ROV operations could have been completed with two or three personnel plus a boat captain. However, the post-trip processing time is much greater for ROV data (6 − 8 hrs per survey) than for that derived from RDS (30 min per survey) and can ultimately result in greater personnel costs. If dive surveys deeper than recreational diving limits are warranted (as our data seem to show for these platform reefs), the use of mixed gases or rebreather technology and the associated equipment additions and advanced training will lessen the difference in startup costs between the two methods. Additional concerns about safety, equipment maintenance, and availability of highly trained personnel often discourage scientists from using this specialized dive equipment (Sieber and Pyle, 2010). Choices related to ROV use are often negatively impacted by the start-up cost for ROV survey equipment, the much greater post-processing time for video, and subsequent data backlogs (Tessier et al., 2005; Murphy and Jenkins, 2010; Pita et al., 2014). However, with ROV fewer field personnel are needed, a permanent video record of the fish community can be maintained, dive time is unlimited, more sites and deeper depths can be surveyed (Lam et al. 2006), and overall risk to personnel is controlled. Because our study indicates that access to deeper depths is important to document inhabitants of these platform reefs in the GOM, we suggest the use of an ROV in addition to diver-based survey methods. The addition of ROV technology such as an Ultra Short Base Line (USBL) positioning system would enhance surveys with more precise location and distance information, ultimately resulting in more accurate density estimates for this habitat type in the GOM.

5. Conclusions

Our data demonstrate that particular care needs to be taken when choosing methods for evaluating differences in communities or specific species, and for Texas platform reefs in particular, recent ROV survey data should be compared to longer term RDS datasets with caution. Not surprisingly, the combination of dive and ROV methodologies seems to provide the most comprehensive community survey perhaps due to a reduction in method bias compared to using a single survey method (e.g., reducing the effects of avoidance to certain technology). Access with an ROV to deeper bottom depths and greater survey time also aids in the detection of more deeply distributed species, although poor visibility at deepest depths may decrease confidence in associated abundance data. For several species of recreational and commercial interest, particularly Red and Vermilion Snapper, our data indicate the ROV method used in this study to be the survey method of choice. With more frequent detection and more conservative abundance estimates for these species, our results document the importance of understanding and evaluating how method choice may influence accuracy of data collected. In the future, the addition of technical divers who can survey deeper depths, as well as incorporating UVC techniques designed to more accurately measure fish density, will increase confidence and clarify discrepancies seen in resulting abundance estimates. Future studies should investigate the extent and direction of associated method bias by a further examination of species distributions on these high-relief platform reefs. In addition, the attraction of certain species to underwater vehicles deserves further investigation. Although our data indicate that both RDS and ROV can be effective means of sampling, researchers should carefully consider costs and inherent method-specific bias particularly for projects regarding the management of certain species. We recommend a combination of methodologies to fully document the communities using these structures.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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