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Comparison of video and traps for detecting reef fishes and quantifying species richness in the continental shelf waters of the southeast USA

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ABSTRACT: The management of reef-associated fish species is challenging due to their life history characteristics and occurrence in rocky reef habitats that are difficult to sample using traditional sampling gears. We compared 2 common sampling gears for reef fish species, chevron traps and underwater video, using 5 yr of comprehensive paired sampling data (N = 7034) collected at reef habitats between North Carolina and Florida along the southeast United States Atlantic continental shelf. Most fish families (80%) and species (85%) were observed significantly more often on video than caught in traps, and only 2 species out of 40 (5%) — black sea bass *Centropristis striata* and bank sea bass *C. ocyurus* — were caught significantly more often in traps than observed on video. Moreover, site-specific species richness was approximately 3–4 times higher on average for video compared to traps. We also used a generalized additive model to show that the ratio of trap-caught species to video-observed species was higher in shallower waters off North and South Carolina, especially when water clarity was low. Results demonstrate that video can be an efficient gear to sample most reef fishes around the world but may provide even greater benefits when paired with traditional sampling gears like traps to leverage the strengths of each gear for estimating relative abundance and species richness with more certainty.

KEY WORDS: Biodiversity · Baited remote underwater video stations · BRUVS · Detection · Camera · Sampling gear · Generalized additive model · GAM · Paired gears

1. INTRODUCTION

Many reef-associated fish species are economically important, providing food, sport, and ecotourism opportunities for people around the world (Coleman et al. 1999, Moberg & Folke 1999), yet are difficult to manage given their life history characteristics. Various reef species on the southeast US continental shelf (SEUS), for example, grow slowly (Wyanski et al. 2000), mature late (Harris et al. 2004), display relatively high site fidelity (McGovern et al. 2005, Bacheler et al. 2021), exhibit complex social structure (Colin 1982), aggregate to spawn (Farmer et al. 2017), and change sex (Harris et al. 2002), all of which render them vulnerable to exploitation (Coleman et al. 1999).

Reef fishes also often inhabit rugose demersal habitats that are challenging to survey using traditional sampling gears. Whereas trawls can often be used to sample fishes on soft-bottom substrates, no comparable sampling gear can be used in rocky, untrawlable habitats. Commonly used sampling gears like bottom

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trawls, gill nets, and longlines often get snagged on rocky and coral reef habitats (Freese et al. 1999, Enrichetti et al. 2019), which damages the fragile sessile organisms growing in these habitats (Hall-Spencer et al. 2002, Mangi & Roberts 2006). Historically, fishes in rocky and coral reef habitats have mostly been surveyed using traps (Munro et al. 1971, Munro 1974, Miller 1990) and hook-and-line sampling (Rudershausen et al. 2008, Vidal et al. 2018), but underwater visual census, fisheries acoustics, and optical gears mounted on underwater vehicles or stationary platforms have become increasingly common in recent years (Willis et al. 2000, Yoklavich et al. 2007, Bacheler et al. 2016a, Rasmuson et al. 2021).

Currently, the 2 most common sampling gears for reef fishes in untrawlable habitats are traps (Collins 1990, Recksiek et al. 1991, Evans & Evans 1996, Jones et al. 2003, Wells et al. 2008, Rudershausen et al. 2010, Bacheler & Smart 2016) and stationary video (Mallet & Pelletier 2014, Aguzzi et al. 2015, Bacheler et al. 2019). However, few studies have compared these 2 sampling gears in a comprehensive manner. Wells et al. (2008) showed that there were differences in the catchability and size selection of red snapper *Lutjanus campechanus* between chevron traps and underwater video in the Gulf of Mexico. Bacheler et al. (2013a) examined the frequency of occurrence of 15 common reef fish species in traps and video in the SEUS, and 11 of these species were observed more frequently on video than caught in traps and 4 species were not significantly different between gears. More recently, Bacheler et al. (2017) compared simultaneous underwater visual census, video, and trap sampling for 7 reef fish species in Florida, and showed that the optimal sampling gear varied among species. The drawbacks of these studies were that sample sizes were small (N < 300), relatively few species were examined, only 1 or 2 yr of data was included in each study, and the spatial footprint of sampling was relatively small.

Here, we expand on previous efforts to conduct a comprehensive comparison of traps and stationary underwater video for sampling reef fishes in the SEUS. In our study, we use 5 yr of paired trap and video sampling data collected over a large geographic area (i.e. ~100 000 km^2) in SEUS continental shelf waters that included large sample sizes and examined dozens of fish families and species to address 4 objectives. Our first objective was to compare the frequency of occurrence of fish families caught in traps to those observed on video, because family-level analyses can be applicable to video surveys occurring around the world where species might be different but many of the same families are represented. The second objective was to conduct the same analysis but at the species level, focusing on species with economic and ecological importance in the SEUS. Our third objective was to examine the relationship between the number of species caught in traps (hereafter, used synonymously with 'species richness in traps') to the number observed on video ('species richness on video') to determine which gear may be most useful for community-level analyses. Our last objective was to determine if the ratio of trap-caught species to video-observed species was invariant across the sampling area or varied by environmental conditions, habitat variables, or space. Our results suggest that using paired sampling gears may be advantageous over any singular gear, which may be applicable to other reef fish species and regions of the world where researchers are, or will be, conducting trap or video surveys.

2. MATERIALS AND METHODS

2.1. Study area

The continental shelf of the SEUS is an expansive area between North Carolina and Florida that varies from 10 km wide off southern Florida to 130 km wide off Georgia (Fig. 1). The primary substrate of the SEUS is sand, but there are patches of natural hardbottom that occur throughout the SEUS (Powles & Barans 1980, Parker et al. 1983, Schobernd & Sedberry 2009, Steward et al. 2022) with which a diverse reef fish community associates (Kendall et al. 2008, Bacheler et al. 2019). These natural hardbottom reefs are highly variable and include rocky ledges, outcrops, boulders and rubble, steep scarps, and flat pavements often covered by a thin veneer of sand (Barans & Henry 1984, Parker & Mays 1998). The dominant hydrographic feature of the region is the Gulf Stream, which flows northward along the continental shelf break (Atkinson et al. 1985) and strongly structures the fauna of the region via its warming effects on water temperature (Whitfield et al. 2014) and circulation effects on transport and recruitment of eggs and larvae (Checkley et al. 1988, Myers & Drinkwater 1989).

2.2. Data collection

We used trap and video data collected by the Southeast Reef Fish Survey (SERFS) for this study. The SERFS is composed of 3 fishery-independent

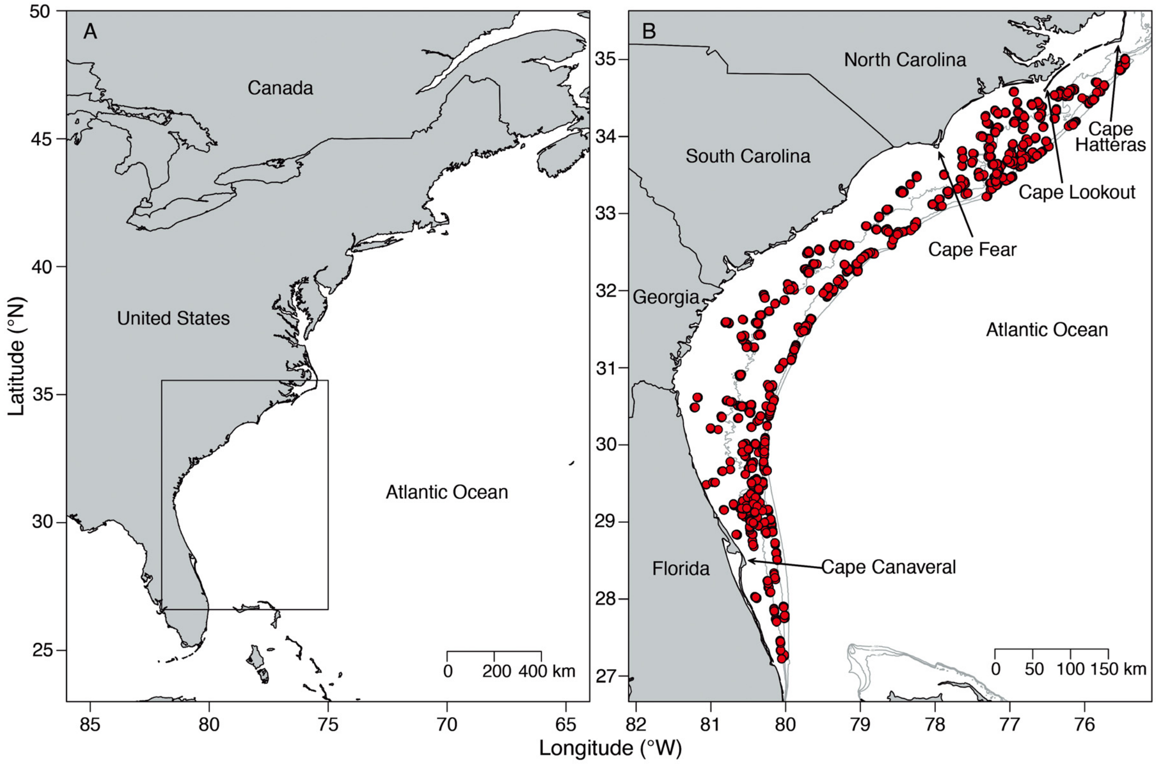

Fig. 1. (A) East Coast of the USA, with the rectangle delineating the study area on the southeast US Atlantic continental shelf. (B) Red points show specific sampling locations for this study in 2015–2019, and overlap in many instances. Gray isobaths indicate 30, 50, and 100 m deep

sampling programs that work together using identical methodologies to survey reef fishes in the SEUS: (1) the Southeast Fishery-Independent Survey, (2) the Marine Resources Monitoring, Assessment, and Prediction program of the South Carolina Department of Natural Resources, and (3) the Southeast Area Monitoring and Assessment Program – South Atlantic that supports the South Carolina Department of Natural Resources. All programs were funded by the National Marine Fisheries Service. We used data from 2015 to 2019 because, during this period, trap and video data were collected by SERFS in a consistent manner, and the spatial and temporal extent of sampling each year was consistent and expansive.

A simple random sampling design was used to select stations for sampling each year. From a sampling frame consisting of approximately 4000 potential sampling stations located on or near hardbottom, approximately 1500 stations were selected for sampling each year. Although most stations sampled in this study were randomly selected (80%), some (17%) were sampled despite not being randomly selected in order to increase efficiency during research cruises and were also included in the analyses. In addition, a small number of new hardbottom stations (3%) were sampled each year based on information from sonar maps, fishing charts, or fishermen, and were included if hardbottom was observed. Sampling occurred during daylight hours on 4 research vessels: RV 'Savannah', RV 'Palmetto', NOAA Ship 'Pisces', and SRVx 'Sand Tiger'.

We quantified the presence (i.e. detection) of various reef fishes using the SERFS paired sampling gear approach, whereby underwater video cameras were attached to chevron fish traps (Fig. 2). Chevron traps were shaped like an arrowhead as viewed from above, constructed from plastic-coated galvanized 2 mm diameter wire mesh (mesh size = 3.4 cm), and measured 1.7 × 1.5 × 0.6 m in size (Collins 1990). The trap mouth opening was approximately 18 cm wide and 45 cm tall, and shaped like an upside-down teardrop (Bacheler et al. 2013b). Each chevron trap was baited with 24 menhaden (*Brevoortia* spp.), 4 on each of 4 stringers inside the trap and 8 loose in the trap. Traps were deployed individually in groups of 6 or fewer, and the minimum distance between any simultaneously soaking traps was 200 m to provide independence among traps (Bacheler et al. 2018, 2022). Target soak time for each trap was 90 min, and traps not fishing correctly (e.g. upside down,

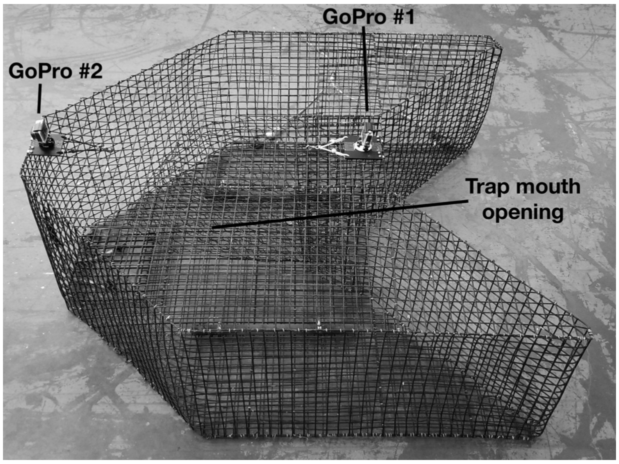

Fig. 2. Chevron trap with 2 attached GoPro Hero 4 underwater video cameras used on the southeast US Atlantic continental shelf in 2015–2019. Note that GoPro #1 is positioned over the trap mouth and GoPro #2 is positioned over the nose of the trap, and both cameras face away from the trap

trap mouth obstructed, bouncing or dragging traps) were excluded from analysis. All fish caught in chevron traps were identified to species, enumerated, weighed, and measured for total length. Trap detection of a taxon implies that at least one individual of that taxon was caught in that particular trap.

Each trap deployed in this study was also outfitted with 2 high-definition cameras. One GoPro Hero 3+/4 was attached over the trap mouth, and one was attached over the trap nose, each looking outward away from the trap (Fig. 2). Fish detections were only evaluated using the camera over the trap mouth, while both cameras were used to quantify substrate and water clarity in opposite directions. For this study, videos were read for the presence of fish taxa over a continuous 20 min segment of video beginning 10 min after the trap landed on the bottom. At least one individual of a particular taxon had to be observed on the 20 min video segment to be scored as a positive video detection for that taxon. Videos that could not be read for any reason (e.g. too dark, obstructed view, files corrupt, full 20 min video was not available) were excluded from analysis. Since the focus of the study was to compare the frequency of occurrence of various species caught in traps to those observed on videos, samples were only included in our analyses if both trap and video samples were valid for a particular station. Valid paired trap and video data included in our analyses are hereafter referred to as 'trap–video samples.'

Comparing frequency of occurrence of fish from cameras attached to chevron traps provides a direct comparison between gears, but the downside is that gears are not independent (see Bacheler et al. 2017 for more details). For instance, it is possible that chevron traps catch some fish soon after deployment that are then unable to be viewed on video, but this is likely a negligible source of bias given that traps often catch a relatively small portion of the fish available at a given site (i.e. low trap catchability; Bacheler et al. 2013b, Coggins et al. 2014). Also, only a 20 min segment of video was read during a 90 min trap soak, so the temporal overlap of sampling was relatively low in our study. Reading the entire 90 min video would have likely increased the frequency of occurrence of species on video, but the costs of video reading would have been much higher.

Characteristics of the water and substrate at each site were also estimated for each trap–video sample. Depth was estimated using ship-board sonar, latitude was determined via a global positioning system, and bottom water temperature was measured for each group of simultaneously deployed traps using a conductivity-temperature-depth cast deployed within 2 m of the seafloor. Percent hardbottom was visually estimated from both video cameras attached to the trap as the percent of the bottom substrate that consisted of hard, consolidated sediment at least 10 cm in diameter. The overall percent hardbottom value for each station was the mean percent hardbottom estimated from the 2 cameras. Substrate relief was visually estimated as the maximum relief of the substrate, scored as low (<0.3 m), moderate (0.3–1.0 m), or high (>1.0 m; Bacheler et al. 2014). Trap soak time was the duration of the trap deployment and ranged from 50 to 150 min. Water clarity was also estimated from each of the 2 videos attached to chevron traps and was scored in 3 qualitative categories: 'poor' if the substrate could not be seen, 'fair' if the substrate but not the horizon could be seen, and 'good' if the substrate and the horizon could be seen.

2.3. Data analysis

We used presence–absence data — i.e. whether a taxon was caught in a trap or observed in a video or not — for all analyses in our study. Presence–absence data is a lower-cost state variable that is often used in place of abundance or density and can be used to infer occupancy rate, which is the proportion of sampling units occupied by a particular species (MacKenzie et al. 2006). In addition to evaluating optimal sampling gears, presence–absence data can shed light on many topics of management importance including changes in a species' spatial distribution

(Steen et al. 2021), invasions by exotic species (Gormley et al. 2011), and metapopulation dynamics (Chandler et al. 2015). In some cases, presence–absence data can also be used as a substitute for population size or abundance, especially over broad areas and for territorial or cryptic species (MacKenzie 2005).

Our first analysis compared the frequency of occurrence of reef fish species at the family level between those observed on video and those caught in traps. There were 2 exceptions to family groupings in our analyses. First, various shark species from numerous families were observed infrequently in our study, so instead of analyzing them separately at the family level, we analyzed them together as a single 'shark' grouping. Second, identification of various flatfishes (order Pleuronectiformes) to the family level was not possible using video, so they were examined together at the order level here. For ease of reference, we subsequently call these 'family' analyses despite the 2 exceptions.

We then used the frequency of occurrence of each family observed on video and caught in traps to calculate the percent increase or decrease on videos compared to traps as:

$$P = \frac{v - t}{t} \times 100 \tag{1}$$

where P is the percent increase or decrease on videos, v is the number of videos in which any individuals from a particular family were observed, and t is the number of traps in which any individuals from a family were caught. We determined statistical significance of potential differences between trap and video frequency of family occurrence using a 2-tailed exact binomial test. An exact binomial test assesses the null hypothesis that frequencies from 2 categories (in our case, trap and video) are equal and was chosen over other tests (e.g. chi-square) due to its ability to handle small sample sizes (Sokal & Rohlf 1995). This and all subsequent analyses were conducted in R version 4.1.1 (R Core Team 2021).

Our second analysis mirrored the first analysis but occurred at the species instead of the family level. Here, we focused on 40 species (across 10 families) that have economical or ecological importance in the SEUS. Note that 2 species of Indo-Pacific lionfish (*Pterois volitans*, *P. miles*) exist in the SEUS and are nearly morphologically identical (Hamner et al. 2007), so they were grouped in our analyses as a single species (*Pterois* spp.). We calculated the percent increase or decrease on videos as above and also used 2-tailed exact binomial tests to determine significance.

To further elucidate the strengths and weaknesses of chevron traps compared to underwater video, our third analysis related the number of fish species caught in traps to the number of fish species observed on video (i.e. paired data). We used a boxplot to illustrate the relationship between traps and videos, showing the median number of species caught in traps for each number of species observed on video. We also fit a locally weighted scatterplot smoothing (i.e. LOWESS) function that predicted mean species richness in traps given specific values of species richness on video.

The efficacy of estimating species richness from traps and video may also vary across space, environmental conditions, or habitat variables. Therefore, our fourth analysis used a generalized additive model (GAM) that related the number of species caught in the trap to various predictor variables. GAMs are semiparametric regression models that can relate a response variable to multiple predictor variables in nonlinear ways, while also incorporating different error distributions (Hastie & Tibshirani 1990, Wood 2006).

We used the number of species caught in traps as the response variable in our GAMs. By itself, this model formulation would simply predict the conditions where more or fewer species were caught in traps, which was not a focus of this study. To specifically address our objective, we also included the number of species observed on video as a predictor variable in the GAM. This model formulation allowed us to standardize for the number of species observed on video and, thus, predict the conditions under which traps do a better or worse job of estimating species richness relative to video. Thus, the response can be interpreted as the ratio of the number of species caught in the traps compared to the numbers being observed on video. We also explored 2 alternative response variables. First, we formulated a model that included the ratio of species caught in traps to those observed on video as a response variable, but that was not an appropriate model formulation because the relationship between these 2 variables was nonlinear (see Section 3 'Results'). Second, we included the number of species caught in the trap as the response variable and the number of species observed on video as an 'offset' predictor variable, but this formulation was similarly rejected due to the nonlinear relationship between trap and video species richness.

Our full GAM model was formulated as:

$$y = s_1(depth) + s_2(lat) + s_3(temp) + s_4(ph) + s_5(soak) + s_6(video) + f_1(clarity) + f_2(relief) \tag{2}$$

where y is the number of species caught in the chevron trap, *depth* is the water depth (m), *lat* is the latitude (°N), *temp* is the bottom water temperature (°C), *ph* is percent hardbottom, *soak* is the trap soak time (min), *video* is the number of species observed on video, *clarity* is the water clarity, *relief* is maximum substrate relief, s_{1-6} are nonparametric smoothing functions, and f_{1-2} are categorical functions. Models were developed using the 'mgcv' library 1.8-36 (Wood 2011).

We evaluated various error distributions and data transformations for our GAM. We compared gamma, negative binomial, Poisson, Tweedie, and Gaussian error distributions, as well as log and fourth-root transformations, using model diagnostics produced by the 'gam.check' function in the 'mgcv' library. The Gaussian distribution without any data transformations was the best fitting error distribution based on visual inspection of the quantile–quantile and various residual plots, and the final model met assumptions of normality and constant variance.

We compared the full GAM to a number of reduced models that contained fewer predictor variables using the Akaike information criterion (AIC; Burnham & Anderson 2002). The benefit of the AIC is that it balances the number of parameters of a model and its log-likelihood, choosing the most parsimonious model in the model set. The model formulation with the lowest AIC value was considered the best model; here, we used ΔAIC values, which were calculated as the difference in AIC values between the best model (ΔAIC = 0.0) and that particular model of interest. Generally, models within 2 AIC units of each other have equal support from the data, and a model more than 2 AIC units lower than others is considered significantly better (Burnham & Anderson 2002).

3. RESULTS

A total of 7034 trap–video samples were collected and analyzed from 2015 to 2019 (Table 1). The number, dates, latitudes, and depths of trap–video samples were very similar among the 5 yr. Each year, sampling occurred from Cape Hatteras, North Carolina, to St. Lucie Inlet, Florida, from approximately 15 to 115 m deep (Fig. 1).

A total of 50 fish families were observed on video, while 29 were caught in chevron traps (Fig. 3). The most commonly observed families on video were Sparidae (N = 5280; 75.1% of videos), Serranidae (N = 5279; 75.0%), Carangidae (N = 4807; 68.3%), Labridae (N = 3883; 55.2%), and Lutjanidae (N = 3863; 54.9%), whereas the most commonly

Table 1. Annual sampling information for trap and video data collected by the Southeast Reef Fish Survey during 2015–2019 on the southeast US Atlantic continental shelf. N: number of paired trap and video samples included in the analyses. Mean values are provided for date (mo/d), latitude (° N), and depth (m); ranges shown in parentheses

Year	N	Mean date (range)	Mean latitude (range)	Mean depth (range)
2015	1385	7/4 (4/21–10/22)	31.9 (27.3–35.0)	38 (16–110)
2016	1399	8/2 (5/4–10/26)	32.2 (27.2–35.0)	41 (17–115)
2017	1420	7/4 (4/26–9/29)	32.0 (27.3–35.0)	40 (15–111)
2018	1314	6/21 (4/25–10/4)	31.9 (27.3–35.0)	40 (16–114)
2019	1516	7/2 (4/30–9/25)	32.1 (27.3–35.0)	40 (16–110)
Overall	7034	7/16 (4/21–10/26)	32.0 (27.2–35.0)	40 (15–115)

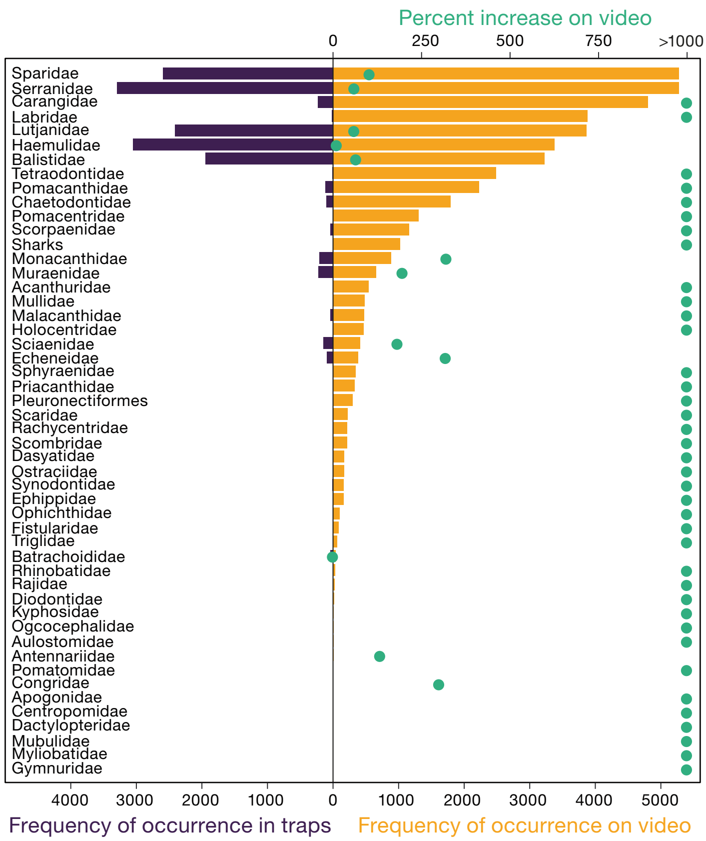

Fig. 3. Frequency of occurrence of fish families from paired video (right of zero, orange bars) and trap (left of zero, dark purple bars) sampling on the southeast US Atlantic continental shelf in 2015–2019. Green points (top axis) shows the percent increase in frequency of occurrence on video compared to traps for all fish families. Note that sharks from all families were grouped into a single 'Sharks' category, and all Pleuronectiformes were grouped at the order level due to identification issues for flatfishes across various families

caught families in traps were Serranidae (N = 3293; 48.6 % of traps), Haemulidae (N = 3054; 43.4 %), Sparidae (N = 2598; 36.9 %), Lutjanidae (N = 2414; 34.3 %), and Balistidae (N = 1949; 27.7 %). Most fish families (40 out of 50; 80 %) were observed significantly more frequently on video than they were caught in traps (2-tailed exact binomial tests: $p < 0.05$), and 38 of the 50 families were more than 1000 % more likely to be observed on video than caught in traps. Of the families with at least 10 observations on video and traps, only Batrachoididae were caught in traps in similar proportion to being seen on video (Fig. 3).

A total of 40 species across 10 families were examined at the species level due to their economic or ecological importance (Fig. 4). Only 4 of these species were observed less frequently on video than they were caught in traps, and only 2 of these species had a statistically significant higher frequency of occurrence in traps than videos: black sea bass *Centropristis striata* and bank sea bass *C. ocyurus*). In contrast, 36 species were observed on video more frequently than they were caught in traps, and all were statistically significant ($p < 0.05$) except blackfin snapper *Lutjanus buccanella* and blueline tilefish *Caulolatilus microps*. Twenty of the 40 species (50 %) were more than 1000 % more likely to be observed on video compared to being caught in traps (Fig. 4). A total of 8 species (21 %) were observed on video but were never caught in the associated traps, most notably goliath grouper *Epinephelus itajara*, yellowtail snapper *Ocyurus chrysurus*, sheepshead *Archosargus probatocephalus*, and queen triggerfish *Balistes vetula*.

There was an asymptotic relationship between the number of species caught in traps and the number of species observed on video (Fig. 5). At low numbers of species observed on video, the numbers of species caught in traps increased linearly but at only about 25–30 % of the number of species observed on videos. Beyond 10 species observed on video, however, the number of species caught in corresponding traps increased very little. For instance, the median number of species caught in traps was 3 if 10 species were observed on video but only increased to 4 when 30 species were observed on video (Fig. 5). The LOWESS smoothed fit corresponded closely to median values from the boxplot (Fig. 5).

The full GAM including all 6 predictor variables was selected by AIC and explained 40.3 % of the model deviance (Table 2). The next best model excluded percent hardbottom but was substantially worse than the full model ($\Delta AIC = 15.6$); all other models were much worse based on AIC values. The response variable was the number of species caught in traps, but the response variable was standardized by the number of species observed on video, meaning that the response can be interpreted as the ratio of the number of species caught in the traps compared to the numbers being observed on video. This response ratio was highest in shallower water (20–35 m deep) and lowest in deeper waters (40–70 m), and it was higher in more northern ar-

Fig. 4. Frequency of occurrence of priority fish species (scientific names color-coded by fish families) from paired video (right of zero, orange bars) and trap (left of zero, dark purple bars) sampling on the southeast US Atlantic continental shelf in 2015–2019. Green points (top axis) show the percent increase in frequency of occurrence on video compared to traps (right of zero) or vice versa (left of zero) for all species

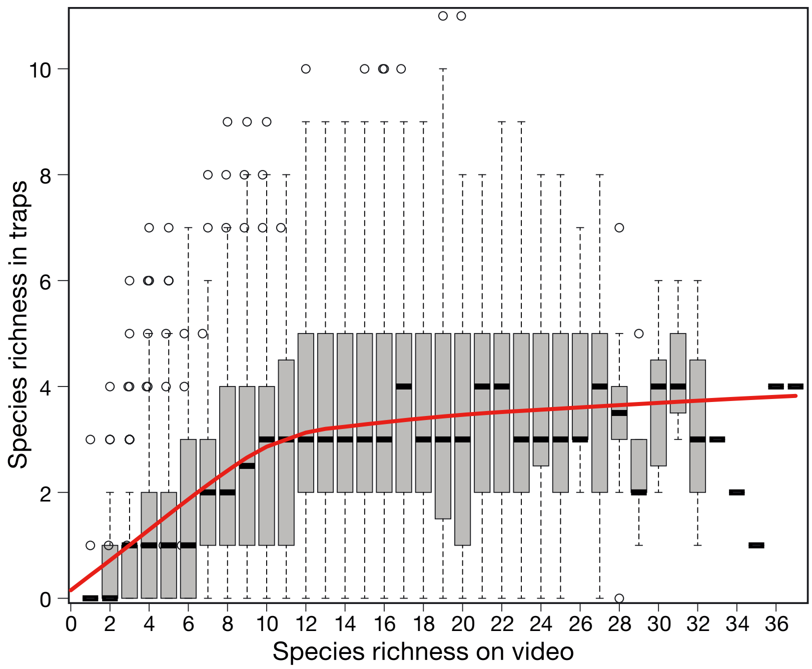

Fig. 5. Boxplot of species richness caught in traps at different levels of species richness observed on video during paired trap–video sampling (N = 7034) on the southeast US Atlantic continental shelf, 2015–2019. Thick horizontal lines are medians, boxes represent the first and third quartiles, whiskers are 1.5 times the interquartile range, and any trap richness values beyond the whiskers are shown by open points. Red line shows a LOWESS (locally weighted scatterplot smoothing) trendline

eas of the study area (i.e. >32° N; North and South Carolina), compared to areas farther south (i.e. Florida; Fig. 6). The ratio of trap to video species was lowest when bottom water temperatures were around 20°C and was slightly higher in colder and warmer waters. The ratio was also slightly higher at lower levels of percent hardbottom (~10% hardbottom) and then decreased at higher levels of percent hardbottom. There was an asymptotic response between the ratio response and the number of species observed on video (Fig. 6), consistent with the raw relationship between the number of species caught in traps and observed on video (Fig. 5). There was also a positive relationship between the ratio response and trap soak time and negative relationships with water clarity and substrate relief, suggesting a higher proportion of species were caught in traps when soak times were longer, water clarity was poor, and substrate relief was low (Fig. 6).

4. DISCUSSION

Fishery-independent survey data is becoming a more important component of fisheries stock assessment and management due to increased regulations that result in the diminished utility of fishery-dependent data. Therefore, determining the most efficient, accurate, and useful fishery-independent sampling gears has become critical for the sustainable management of fish (Murphy & Jenkins 2010). We compared 2 of the most commonly used sampling gears for reef fishes, traps and video, and found that the frequency of occurrence for most fish families and species was higher for video than traps in the SEUS. Moreover, site-specific species richness

Table 2. Model selection for generalized additive model relating the number of species caught in traps to predictor variables, based on trap and video data collected by the Southeast Reef Fish Survey in 2015–2019. Degrees of freedom are shown for factor (f) terms; estimated degrees of freedom are shown for nonparametric smoothed terms (s). DE%: percent deviance explained by the model; ΔAIC: delta Akaike information criterion (best model = 0.0); ex: parameter excluded from model; *depth*: water depth (m); *lat*: latitude (° N); *temp*: bottom water temperature (°C); *ph*: percent hardbottom; *soak*: trap soak time (min); *video*: number of species observed on video; *clarity*: water clarity; *relief*: maximum substrate relief. **: significance at an alpha value of 0.001; ***: significance at 0.0001

Model	DE%	ΔAIC	s_1(*depth*)	s_2(*lat*)	s_3(*temp*)	s_4(*ph*)	s_5(*soak*)	s_6(*video*)	f_1(*clarity*)	f_2(*relief*)
Full	40.3	0.0	7.5***	8.6***	5.2***	5.4***	1.0***	4.3***	2***	2***
Full – *ph*	40.1	15.6	7.5***	8.6***	5.3***	ex	1.0***	4.5***	2***	2***
Full - *relief*	40.1	21.7	7.4***	8.6***	5.1***	5.7***	1.0***	4.5***	2***	ex
Full – *soak*	40.0	29.1	7.5***	8.7***	5.2***	5.6***	ex	4.2***	2***	2***
Full – *temp*	39.6	74.0	7.5***	8.6***	ex	5.7***	1.0***	4.6***	2***	2***
Full – *clarity*	39.3	109.8	7.4***	8.6***	6.3***	5.3**	1.0***	4.2***	ex	2***
Full – *lat*	35.7	515.8	7.5***	ex	7.9***	4.3**	5.7***	4.3***	2***	2***
Full – *depth*	34.0	694.0	ex	8.8***	8.1***	3.3***	1.0***	4.6***	2***	2***
Full – *video*	33.1	797.9	7.5***	8.7***	8.3***	8.4***	1.1***	ex	2***	2***

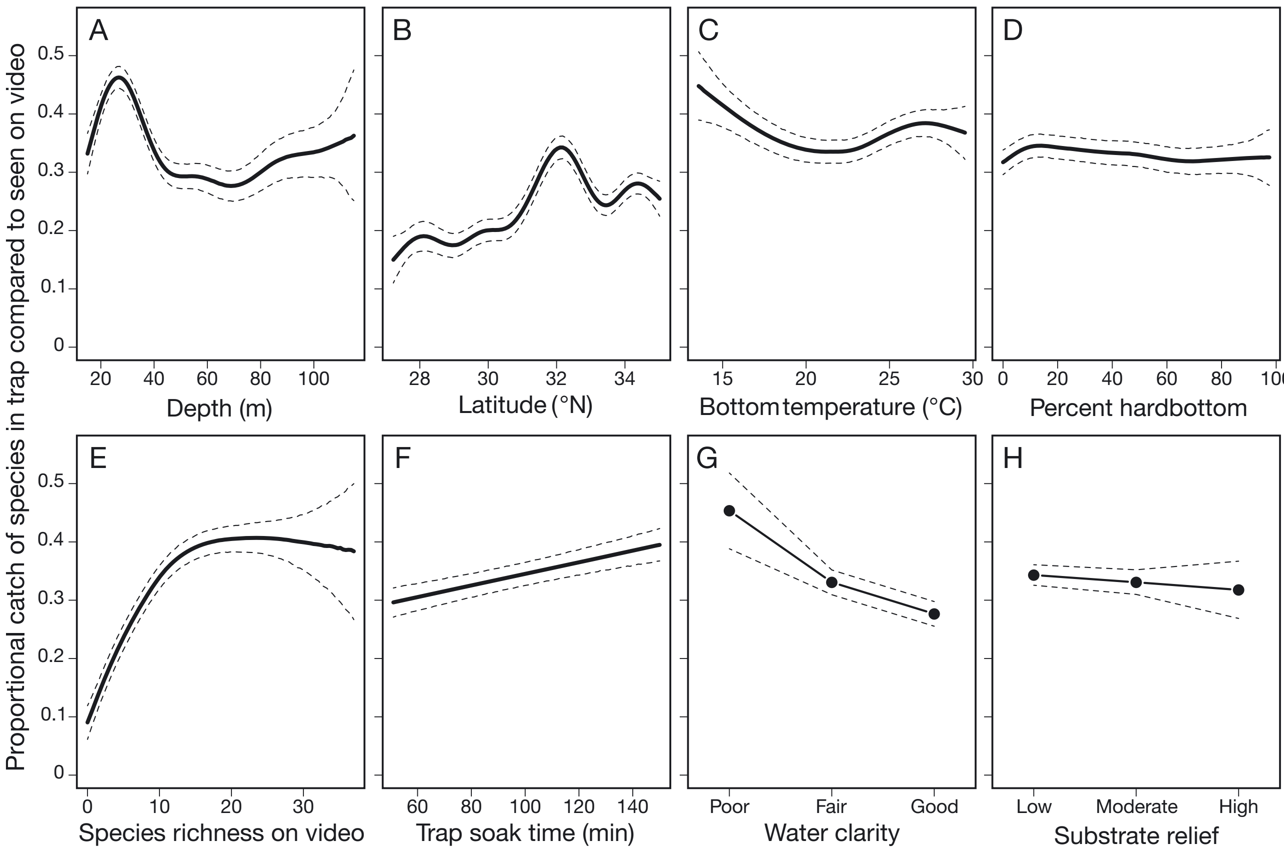

Fig. 6. Relationships between the number of species caught in traps (standardized by the number of species observed on video) and depth, latitude, bottom water temperature, percent hardbottom, species richness on video, trap soak time, water clarity, and substrate relief from generalized additive models built on paired trap–video sampling on the southeast US Atlantic continental shelf, 2015–2019. Solid line is the mean effect, and dashed lines are 95 % confidence intervals. Higher values indicate traps are catching a higher percentage of species than observed on video; lower values indicate traps are catching a lower percentage of species than observed on video

was approximately 3–4 times higher on average for video compared to traps. These results are consistent with previous work that has shown that video tends to be less selective than many other sampling gears (Willis et al. 2000, Cappo et al. 2004, Morrison & Carbines 2006, Harvey et al. 2012, Bacheler et al. 2017).

There is a strong relationship between the proportion of zero catches from a sampling gear and the resulting uncertainty from an index of abundance (Maunder & Punt 2004, Kimura & Somerton 2006). In our study, a vast majority of reef fish families (80 %) and species (85 %) were observed significantly more often on video than caught in traps using continuous reads, meaning video-based indices of abundance may be more precise than trap-based indices due to the lower rate of zeroes. In addition, we compared the frequency of occurrence of fish on video from a 20 min interval of time to traps that soaked for 90 min; frequency of occurrence on video would have been higher had a longer video segment been read and lower if particular frames within the 20 min interval were read (Bacheler & Shertzer 2015). There are additional benefits of using video to survey reef fish over other gears, including the non-extractive nature of video sampling, the ability to sample deeper water more easily than divers and more cheaply than autonomous underwater vehicles (Langlois et al. 2010), providing a permanent record that can be reviewed by multiple readers, and allowing for habitat and behavioral information to be collected (He 2003, Silveira et al. 2003).

Video also appears to be more useful in making inferences about patterns in reef fish biodiversity than traps (Harvey et al. 2012). Consistent with previous work, our baited traps mostly caught mobile predator and scavenger species (Robichaud et al. 2000, Bacheler & Smart 2016, Bacheler et al. 2017). When highly selective trap data is used to make inferences about patterns in fish biodiversity, results only pertain to those select species that are effectively sampled by traps, as described by Bacheler & Smart (2016). Because video often observes a wider variety of species across most or all major functional groups at a site (Cappo et al. 2006, Harvey et al. 2012), observed patterns in biodiversity from video likely more closely reflect true patterns in the fish community compared to traps (Klibansky et al. 2017).

Despite these advantages, using video to sample reef fishes also has some drawbacks. First, video gears do not provide biological samples (e.g. otoliths, reproductive histology, DNA) upon which most age-based stock assessments depend. Second, video cameras and the subsequent video reading can be expensive, although the cost of video cameras has declined in recent years (Langlois et al. 2010). Third, fluctuating environmental conditions like water clarity and current direction can influence video detection (Bacheler et al. 2014). Fourth, stationary video tends to have a more limited view of the seafloor and surrounding habitat and fishes compared to towed cameras, remotely operated vehicles, or divers. Fifth, video can miss small, cryptic, or camouflaged taxa that are present at a site (Mallet et al. 2014, Bacheler et al. 2017), such as Batrachoididae in the present study. And last, counting fish on videos takes time, which can delay video data availability compared to traps.

There are also some distinct advantages of using traps to monitor reef fishes. For some reef fish species, their frequency of occurrence in traps is higher than on video (Wells et al. 2008), as we found for *Centropristis striata* and *C. ocyurus*. Moreover, previous work suggests that the catch of black sea bass in chevron traps is proportional to their site abundance (Bacheler et al. 2013b, Shertzer et al. 2016), which is not always the case for video data, depending on the video reading approach (Schobernd et al. 2014). Other advantages to using traps include the ability to positively identify all individuals caught to the species level and opportunities for extraction of biological samples from collected fish. Although trap catches likely do not reflect the entire fish community (Harvey et al. 2012), traps may be a useful survey gear to assess the relative abundance and biology of select fish species that are strongly attracted to bait (Bacheler et al. 2017).

Given that no sampling gear is able to perfectly sample the entire reef fish community, information from multiple or paired sampling gears may be more advantageous than any single sampling gear. For instance, pairing environmental DNA sampling with traditional baited video provided a broader understanding of the fish community in Western Australia than either approach alone (Stat et al. 2019). Adding video cameras to trawls allowed for the documentation of fine-scale habitat use of various species as well as species overlap (Rosen et al. 2013). Using video in conjunction with traditional sampling gears can not only increase detection of important species but can also be used to estimate and account for imperfect detection of either gear (Bacheler et al. 2014, Coggins et al. 2014, Bacheler & Shertzer 2020). Improved indices of abundance are possible when data from multiple gears can be combined in a statistical framework that accounts for imperfect detection (Gwinn et al. 2019). The advantages of combining gears to provide more comprehensive data, like adding video to traditional sampling gears, often far outweigh the additional cost of deploying a second gear.

Although more species were observed on video than caught in traps, the relative effectiveness of traps and video varied across the study area, environmental conditions, taxa, and habitat. We examined the ratio of trap-caught species to video-observed species, and while that ratio was almost always less than 1 (indicating more species were seen on video than caught in traps), the ratio tended to be higher than average in shallower water in North and South Carolina and lower in deeper water further south. These spatial trends may be the result of the relative proportion of trap-attracted and trap-shy species across the study area. For instance, black sea bass, bank sea bass, tomtate *Haemulon aurolineatum*, white grunt *H. plumieri*, and sand perch *Diplectrum formosum* are all strongly attracted to baited traps and, within our study area, tend to be most commonly found inshore in North and South Carolina (Bacheler et al. 2016b, 2019). Additionally, these inshore areas of North and South Carolina tend to have reduced water clarity compared to other areas, potentially reducing video detectability. Areas further south and deeper continental shelf-break habitats tend to be dominated by smaller tropical species that are less attracted to baited traps (Whitfield et al. 2014, Bacheler et al. 2019). Furthermore, the ratio of trap to video species increased when soak times were longer (because more species were caught in the traps, increasing the numerator) and water clarity was poor (because fewer species were observed on video, decreasing the denominator). Generally, demersal predatory or scavenging taxa at higher trophic levels were more likely to be caught in traps than pelagic or lower trophic level species.

There were some drawbacks of our study. First, we analyzed data from paired trap and video gears because comparisons were direct, but the downside is that the 2 gears were not independent. The main potential concern is that chevron traps may catch fish that are then unable to be viewed on video. While it is unlikely in the current study that chevron traps frequently caught fish that were then unable to be viewed on video because video reading occurred early in the trap soak, if it did occur, then the frequency of occurrence of fish on video may be underestimated. Second, in our GAM analysis, the ratio of trap-caught species compared to video-observed species was affected by factors influencing the trapping efficiency (the numerator) but also the efficiency of video sampling (the denominator). Thus, when the ratio increased (for instance, in poor water clarity), it is impossible to determine if it was due to increased trapping efficiency or decreased video efficiency, or both.

We have provided in the current study a broad comparison of chevron traps and underwater video to detect reef fish species in the SEUS, and demonstrated that frequency of occurrence for most species and families was higher on video compared to chevron traps. There are a number of reef fish surveys around the world that have transitioned to solely using video to survey fishes (e.g. De Vos et al. 2014, Aguzzi et al. 2015, Amin et al. 2017). The use of video to monitor reef fish species in untrawlable habitats, alone or in tandem with other sampling gears, is quickly becoming the preeminent sampling approach. Our results show that it may be advantageous to pair sampling gears, such as attaching video to traditional sampling gears, to leverage information from both gears to improve our understanding of fish detection and patterns in relative abundance (e.g. Gwinn et al. 2019).

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