CSIRO PUBLISHING

Marine and Freshwater Research
http://dx.doi.org/10.1071/MF13230

Use of underwater video to assess freshwater fish populations in dense submersed aquatic vegetation

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Abstract. Underwater video cameras (UVC) provide a non-lethal technique to sample fish in dense submersed aquatic vegetation. Fish often inhabit densely vegetated areas, but deficiencies of most sampling gears bias relative abundance estimates that inform fisheries management. This study developed methods using UVC to estimate relative abundance in dense vegetation using three experimental ponds covered with surface-matted hydrilla (*Hydrilla verticillata*) stocked at different densities of *Lepomis* spp. and largemouth bass (*Micropterus salmoides*). We conducted UVC point counts over 13 weeks to measure relative fish abundance and occurrence from video analysis. Ponds were then drained to obtain true fish densities. In total, fish were detected in 55% of all counts and juvenile and adult *Lepomis* spp. and largemouth bass were enumerated. End-of-season true fish densities ranged across ponds (from 52 to 37 000 fish ha^{-1}). Additionally, pond 2's true density changed substantially from 370 to 12 300 fish ha^{-1}. True population size was accurately reflected in differences in estimated relative abundances obtained from fish counts, as in pond 2 where mean fish counts increased from 0.10 in week 1 to 2.33 by week 13. Underwater video accurately and precisely quantified relative abundance at naturally-occurring fish densities, but this success was reduced at low densities.

Additional keywords: complex habitats, fish sampling, *Hydrilla verticillata*, invasive aquatic plants, largemouth bass, sunfish, video cameras.

Received 27 May 2013, accepted 17 March 2014, published online 29 October 2014

Introduction

Habitat complexity shapes the structure of food webs, mediates predator–prey dynamics, provides refuge for young animals, and fosters increased biodiversity through niche partitioning (Schoener 1974; Gorman and Karr 1978; Crowder and Cooper 1982). Submersed aquatic vegetation functions as both ecologically critical and structurally complex habitat (Rozas and Odum 1988; Dibble *et al.* 1996). Across many life-history stages, fishes (including eggs) utilise submersed aquatic vegetation as food, as a substratum, or as refuge from predation (Crowder and Cooper 1982; Dewey *et al.* 1997). The variety of these uses makes submersed aquatic vegetation habitats highly important for both fishery and conservation concerns.

The complex habitats that fish inhabit are often difficult to access, creating unique challenges for sampling strategies aimed to evaluate fish habitat utilisation and estimate population trends for the management of freshwater fishes and their environments. Consequently, sampling efficiency and variability in sampling gear efficiencies can make quantitative assessments of animal occurrence and abundance in structurally complex habitats difficult (Bayley and Austen 2002; Gu and Swihart 2004). These difficulties often limit the understanding of fish abundance and utilisation of habitats (Bayley and Austen 2002). Many studies have used traditional or modified gears to sample fish communities in submersed aquatic vegetation habitats including pop nets, seines, electrofishing, rotenone and trawls (Dewey *et al.* 1989; Tate *et al.* 2003; Dembkowski *et al.* 2012). However, the capture probabilities of these sampling gears are often unknown and are influenced by habitat cover and, thus, it is important to understand how capture probability varies with gear tactics, gear type and habitat structure. Sampling problems become particularly evident in dense or invasive submersed aquatic vegetation species. *Hydrilla verticillata* is an invasive aquatic plant that became established in the US south-east since the 1960s and can

form dense surface canopies and attain high biomass that prevents effective fish sampling with either passive or active gear types (Langeland 1996).

The lack of effective methods for sampling fish in submersed aquatic vegetation environments means current knowledge of fish–plant interactions is coarse and simplistic (Dibble *et al.* 1996). Conservation strategies often assume that dense submersed aquatic vegetation reduces dissolved oxygen owing to high plant respiration rates at night, therefore limiting fish utilisation and available habitat (Miranda *et al.* 2000). Aquatic plant management actions (e.g. herbicides) are often applied at least in part under the assumption that fish habitat will be improved, but there is a need to evaluate fish habitat use and population sizes within dense submersed aquatic vegetation habitats to understand the validity of these assumptions. Given the pervasiveness and spread of non-native and invasive submersed aquatic vegetation plants affecting aquatic ecosystems globally, managers need better information (e.g. increased statistical power) for fish samples obtained within these ecologically important habitats.

Cameras have been used to estimate animal abundance in fish and wildlife studies (Karanth 1995; reviewed in Cappo *et al.* 2007). Increasingly, marine studies utilise underwater video cameras (UVC) as a means to obtain count data and behaviour of fish communities as these methods present a viable, non-lethal alternative to traditional fish sampling techniques useful in a variety of habitats (Priede and Merrett 1998; Cappo *et al.* 2004; Sheehan *et al.* 2010; Favaro *et al.* 2012). Currently, the use of UVC to sample fish in freshwater ecosystems has been limited to behavioural and species richness studies, not population-level analyses or evaluations of their utility in submersed aquatic vegetation habitats (Chidami *et al.* 2007; Martin and Irwin 2010; Ebner and Morgan 2013). Given this knowledge-gap, our objectives were to: (1) develop methods for use of UVC to estimate fish abundance in dense submersed aquatic vegetation; (2) evaluate how these methods performed with manipulations of fish populations in experimental ponds; and (3) discuss how these methodologies can be broadly applied to freshwater fisheries management.

Materials and methods

Study site

To evaluate the efficacy of the UVC to sample fish in dense submersed aquatic vegetation, we used three rectangular 0.405 ha experimental ponds (maximum depth 2.4 m, average depth 1.7 m) located at a United States Geological Survey facility in Gainesville, Florida during summer 2011. We manipulated fish densities among ponds by varying stocking densities. In June 2011, pond 1 was stocked at 185.2 fish ha^{-1} (75 adult *Lepomis* spp. total stocked individuals >140 mm TL total length) composed of bluegill (*L. macrochirus*) and redear sunfish (*L. microlophus*), pond 2 was stocked at 370.3 fish ha^{-1} (total of 150 adult *Lepomis* spp. stocked), and pond 3 had an established multi-year fish community assumed to be at carrying capacity with *Lepomis* spp. and largemouth bass (*Micropterus salmoides*). The size distribution and species composition were similar for fish stocked into ponds 1 and 2. Individuals of each *Lepomis* spp. between 140 mm and 180 mm TL were grouped and stocked at equal proportions between ponds 1 and 2 (i.e. for every one bluegill 140–180 mm TL in pond 1, we placed two bluegill in pond 2). Individuals >180 mm TL from both species were grouped and stocked at equal proportions between the two ponds as well. Large *Lepomis* spp. were grouped separately as these individuals were likely more fecund and we aimed to distribute reproductive potential proportionally across both ponds. Bluegill comprised 70–80% of all stocked fish across both size groups and both ponds while redear sunfish comprised the rest. All three ponds were aerated continuously near the deepest portion of the pond (the 'catch basin') to prevent fish kills, and all contained surface matted (i.e. ~100% coverage) hydrilla by mid-June. Though fish may alter their behaviour due to the presence of the aerator, any behavioural biases associated with continuous aeration would be equal across all ponds and would not alter the inference gained from the random locations of the UVC point counts. Sampling occurred from July to October 2011 and only during daylight hours (9:00 to 17:00) to ensure sufficient light for the UVC.

We drained all ponds in October 2011 to estimate the true total fish population size at the end of the pond manipulations. As the ponds drained, most fish swam into and collected at the 'catch basin' (~5.0 m^2 in area), which is the deepest portion of the pond and has a concrete area adjacent to the drain that screens the fish from outflowing water. We collected and counted all fish in the catch basin with hand nets after the ponds completely drained. We then walked the perimeter of the pond to find and collect all visible *Lepomis* spp. >140 mm TL to recapture the originally stocked fish, assuming that no age-0 *Lepomis* spp. grew to >140 mm TL in this four month period. During the perimeter walk, we estimated the area of the pond outside of the catch basin that still held any fish, including fish >140 mm TL. We then randomly placed 25–30 quadrats (0.25 m^2) to sample fish in this area of the pond outside of the catch basin. All fish that were within a quadrat were collected by carefully picking through all the hydrilla within that quadrat. We identified fish by taxon (e.g. *Lepomis* spp., juvenile largemouth bass or adult largemouth bass) and analysed quadrat samples in a negative binomial log-likelihood structure to obtain maximum likelihood estimates (MLE) of the total fish population outside the catch basin. In this structure, the total numbers of fish outside the catch basin is explained by the distribution of fish in the quadrat counts and the ratio of the area sampled by quadrat (0.25 m^2) to the area of the drained-pond that still held fish. We calculated 95% confidence intervals for the MLE for each pond using $\pm$ the *t* statistic times the standard error for the population estimate. We then added the MLE for the total numbers of fish outside of the catch basin to the total numbers of fish removed from the catch basin (a known and certain constant) to obtain a final estimate of total fish population size per taxa in each pond with associated 95% confidence intervals coming from the uncertainty from the MLE of the total fish population outside the catch basin. A subsample of the collected fish from the drained-ponds were individually weighed to help convert from fish density to fish biomass to compare with other studies of fish population sizes on Florida lakes.

Materials

We used a sampling system composed of a UVC and a mini digital-video-recorder (DVR) powered by a 12-V DC marine

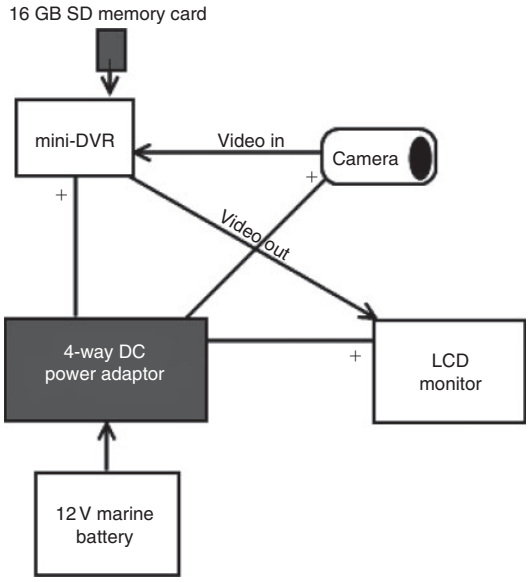

Fig. 1. Schematic design for using the underwater video camera (UVC) system to sample fish in dense submersed vegetation. A 12-V DC marine battery powers the system. Power adaptor connections to the digital video recorder (DVR), UVC, and video monitor are indicated with a + symbol. The 16 gigabyte memory card is used to store video recordings from the DVR. Video cables connect the camera to the DVR, and connect the DVR to the video monitor.

Table 1. Relative costs of equipment needed to build underwater video camera system

This setup is for one complete sampling unit with the number needed noted parentheses. Multiple cameras could be added given subsequent increases to power adaptor, DVR capabilities and cables

Equipment	Price ($, USD)
Camera	250
15.2 m cable length[A]	25
LCD Video Monitor	50
12-V Marine Battery	200
4-way Socket Adaptor	10
mini-DVR	65
16GB s.d. memory card	30
RCA video cable (2)	5
Total	635

[A]15.2 m cable length is standard, additional lengths available.

battery (Fig. 1; Table 1). The single-lens UVC model SMM-50-C (Seaview, Inc.; Tampa, FL; www.seaview.com) is relatively inexpensive (Table 1), 85° wide-angle, colour cameras powered by DC or AC current that provide high resolution (540 lines of resolution) video images in turbid and low-light habitats (up to 0.005 lx with internal infrared (IR) light emitting diodes (LED) turned off). The SMM-50-C automatically alternates between colour (in abundant light) and black-and-white (in low light) signals to maintain optimal video imaging. Colour video allowed fish to be more easily distinguished during fish identifications. We created a mount around the video cable and UVC to orient the camera horizontally (i.e. the camera view was parallel to lake-bottom; Fig. 2). The UVC, DVR and video monitor (liquid-crystal display (LCD)), were powered from the single 12-V battery using a four-way socket power adapter. All equipment (Table 1) was placed within a waterproof case inside a 3.6 m flat-bottom boat.

Field procedures

Fish were counted using 10–20 point counts every two weeks at random locations within each pond. The starting location for each pond in each sample week was also random; for example, if we started collections near the aerator in week 1, we might start week 3 sampling at the other end of the pond. For each video point count, we lowered the UVC from the flat-bottom boat (Fig. 2) and through the submersed aquatic vegetation canopy layer. The dense weight (1.8 kg) and small profile of the UVC allowed the camera to descend with few snags through the dense vegetation, thereby reducing habitat disturbances. The canopy thickness varied spatially, and therefore the depth of UVC deployment could also vary at each site. Our objective was to place the UVC below the hydrilla canopy and this occurred most often at 1.5 m in depth. Before recording, we used the live-feed from the video display to ensure the UVC viewpoint was oriented away from the flat-bottom boat and was not overly obstructed by vegetation. We then used a clamp to hold the video cable against the side of the boat at that depth and began recording video for 10 min per point count on a 16 gigabyte secure digital memory card (maximum storage capacity of 4 h of video recording). We used this sampling design to capture presence–absence (hereafter called occurrence) and count fish beneath the hydrilla surface canopy layer within each pond (Fig. 2). To quantify plant biomass, we used a boat-based vertical rake to sample plant biomass (kg m^{-2} in dryweight) from seven random sites in each pond every month (Johnson and Newman 2011).

Video and statistical analyses

We recorded fish occurrence and counted fish from video analysis at each point count. We used the video playback software GOM media player (http://player.gomlab.com/eng/download/) and analysed each 10 min video in 30 s intervals. This breakdown allowed longer video counts to be assessed quickly (playback speed ~1.5×) such that we could return to an interval without reanalysing the entire 10 min video. We identified fish down to the lowest taxonomic level (e.g. functional group, family, or species). Video saturation, contrast, and brightness could be adjusted in the media player to optimise our ability to enumerate fish and aid in identifications.

Fish occurrence and abundance measures were adapted from previous video studies (Priede and Merrett 1998; Cappo *et al.* 2004; Gledhill *et al.* 2005). If fish were observed at any time in 10 min, we noted fish as present, otherwise fish were considered absent. When fish were present we counted the maximum number of individuals in view at any one time within every 30 s interval. To prevent double-counting of fish within that sampling site, we used the $MaxN_{species}$ statistic as the maximum number of individuals of a taxa onscreen at any one time during the point count (Cappo *et al.* 2004; alternatively called *mincount* in Gledhill *et al.* (2005) and *npeak* in Priede and Merrett (1998)). We utilised generalised linear models (GLM; $\alpha = 0.05$) in Program R to evaluate whether occurrence and *MaxN* metrics

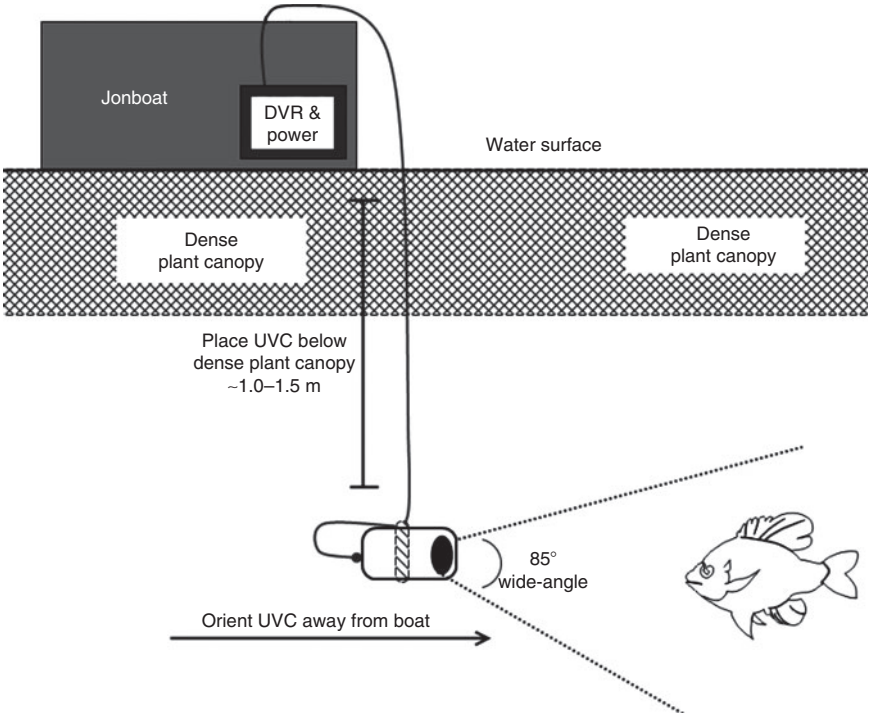

Fig. 2. Diagram for underwater camera deployment to sample fish in dense aquatic plants.

varied between the ponds and through the sampling season with the sample week as the predictor covariate (R Development Core Team 2012). Binomial and negative-binomial regressions were used to test whether fish occurrence and fish counts varied among the ponds. We used logit-transformations to analyse fish occurrence data with the binomial GLM and analysed fish count data with a negative-binomial GLM with the 'glm.nb' function in Program R using the 'CAR' and 'MASS' statistical packages (Venables and Ripley 2002; Fox and Weisberg 2011). Though animal count data from repeated single-observer approaches is often autocorrelated (potentially from double-counting mobile individuals) autocorrelation does not inherently bias results, increase type I error, nor lead to false rejections of the null hypothesis in regression analyses fit with least square residuals (Diniz-Filho *et al.* 2003). We assumed the random locations and starting point of the count locations during each sample week minimised autocorrelation. We chose the negative-binomial regression due to overdispersion in the count data when using the Poisson distribution in initial analyses and the negative-binomial's relaxed constraints on violations of independence in sample replicates (White and Bennetts 1996). The negative-binomial is often utilised to analyse population trajectories in animal count data in such an approach (Link and Sauer 1997; Link and Sauer 1998).

We conducted a nonparametric bootstrap on fish count data at the start and end of the sample season to evaluate the efficacy in the UVC to detect known differences in the fish populations. Each bootstrap simulation (10 000 iterations total) randomly resampled the fish count data from the first sample week and the last two sample weeks at different intensities, from 10 to 30 samples. This assumed that the last two sample weeks (weeks 11 and 13) in each pond held similar true fish densities and the observed count data was representative of the true state of the fish populations (Hilborn and Mangel 1997). Bootstrapping the data in this way allowed us to determine whether known population differences could be detected with the UVC, and if so, the sample numbers needed to determine that difference and gives an inference on statistical power. After each iteration, a one-way or two-way (depending on the ponds tested) Student's *t*-test was conducted on the bootstrapped data to evaluate whether significant differences in the count data between ponds were detected. We tested the assumption that pond 3 had a consistently high fish population at carrying capacity by comparing the UVC fish population estimates in week 1 (where the true population size was unknown) to the UVC fish population estimates in weeks 11 and 13 (where the true population size was known). For each sample intensity scenario, we recorded the 2.5th, 50th, and 97.5th percentile of the mean UVC counts per iteration and the percentage of the Student's *t*-test to come up significant ($\alpha = 0.05$).

We measured habitat complexity for each video count by counting the number of visible plant stems (i.e. a relative stem count) following a horizontal grid from a single-frame captured from the video. If fish were present, habitat complexity was measured around the time interval where the maximum number of fish occurred; if fish were absent, habitat complexity was measured at a time interval that best represented the 10 min video. We used analysis of variance (ANOVA) to evaluate whether habitat complexity and plant biomass (see above) varied between the ponds, and a linear GLM to test whether they changed through the sampling season with sample week as the predictor covariate.

Fish length could not be estimated from video. This was because the single-lens UVC lacks depth distance measurements

Table 2. Numbers of fish collected from netting fish at each ponds' catch basin and from quadrat samples for the area outside the catch basin that still contained fish after pond drainings

The combination of fish caught from the catch basin and maximum likelihood estimates for total numbers of fish outside the catch basin (derived from the quadrat count distribution and the ratio of area sampled to area still containing fish) contributed to total population estimates. The area still containing fish for each pond varied with pond 1's area at 203 m^2, pond 2 at 675 m^2, and pond 3 at 1350 m^2

	Pond 1	Pond 2	Pond 3	
	Lepomis	*Lepomis*	Largemouth bass	*Lepomis*
Numbers of fish collected from catch basin	21	2467	53	3813
Numbers of fish in quadrat	Number of quadrat counts with fish			
0	25	20	27	14
1	–	3	2	2
2	–	–	1	6
3	–	–	–	3
4	–	–	–	1
5	–	1	–	1
6	–	–	–	1
7	–	–	–	–
8	–	–	–	–
9	–	–	–	1
10	–	–	–	–
11	–	–	–	–
12	–	–	–	–
13	–	–	–	1
14	–	–	–	–
15	–	–	–	–
16	–	1	–	–

meaning that a small fish closer to the screen can appear the same size as a larger fish further from the screen which differs from the use of stereo-video techniques to measure fish community size structure (Watson *et al.* 2005). However, we were able to qualify some individuals as either juveniles or adults based on morphological characteristics that vary across ages including: colour, length-to-height ratios and girth.

The boat was manoeuvred around the pond with paddles and the boat-based observations were assumed to have a very low impact on fish behaviour (Graham and Cooke 2008). Individuals in the Centarchidae family, including *Lepomis* spp., respond differently to novel situations such as the presence of paddles or cameras (Graham and Cooke 2008; Wilson and Godin 2009). Populations can be composed of a portion of 'bold' individuals that respond positively to novel items, while others can be composed of 'shy' individuals that respond with avoidance (Wilson and Godin 2009). We tested whether the number of fish counted per time interval was influenced by time interval to evaluate potential gear or behavioural biases. If fish were to avoid the gear at the time of deployment we would expect the number of fish observed on video to be biased towards the end of the video recording as they acclimate to the camera and return to normal behaviours. If fish were attracted to the gear, this would be biased to the beginning of the video sample and we assumed that any biases in *Lepomis* spp. could be detected within 10 min of video recording. The number of fish counted per 30 s time interval per point count was evaluated with a linear regression with time interval as the predictor covariate. In this model we removed all *MaxN* counts that had >15 fish ($n = 1$) to reduce the bias of outlier samples. We also tested the significance of the null linear regression which would indicate that the slope of the fish counts across the video duration is flat (i.e. there is no bias over the video duration). We then evaluated these two models based on Akaike's Information Criterion (AIC; Akaike 1974). Models with ΔAIC value of <8.0 have support and <2.0 have substantial support and are considered equivalent at explaining the data; if models were equivalent (i.e. the two models are differentiated by <2.0 ΔAIC) we selected the most parsimonious model among them (Burnham and Anderson 2004).

Results

Fish density varied among the ponds based on fish collections from the catch basin and quadrat sampling from the drained-ponds. The areas outside the catch basin still containing fish in each drained-pond (i.e. the total area sampled with quadrats) also varied among ponds with pond 1's area at 203 m^2, pond 2's at 675 m^2, and pond 3's at 1350 m^2. Fish density in pond 1 started at 75 adult *Lepomis* spp., and we physically recovered 21 adult fish (14 bluegill and 7 redear sunfish all >140 mm TL) with zero recruitment of young fish in this pond (Table 2). Fish abundance in Pond 2 started at 150 adult *Lepomis* spp. with 66 adult *Lepomis* spp. (54 bluegill and 12 redear sunfish all >140 mm TL) recovered at the end, 2467 fish recovered in the catch basin and a total population size estimated (i.e. the combination of catch basin totals added to quadrat MLE population size) at just over 5000 fish (95% CI 4027–6100) owing to large recruitment of young fish, particularly redear sunfish which comprised >99% of all recruits. Overall, the adult fish stocked into pond 1 declined by 72% (54 of 75 stocked fish) while adult fish in pond 2 declined by 56% (84 of 150 stocked fish) during the course of the study due

to either natural mortality or adult fish remaining unsampled in the area outside the catch basin after the pond draining. In pond 3, we recovered 3813 *Lepomis* spp. and 53 largemouth bass from the catch basin. Total fish population size in pond 3 (assumed to be relatively stable and at carrying capacity) ended with an estimated 15 000 *Lepomis* spp. (95% CI 11 874–17 354, with bluegill and redear sunfish comprising 79% and 16% of all recovered adults respectively), three adult largemouth bass (fish >300 mm TL) and >700 juvenile largemouth bass (fish >80 mm TL; 95% CI 67–1478 fish). A subsample of 479 fish from across the ponds were weighed to get a mean of 0.009 kg individual^{-1}. As largemouth bass were present in low densities in only one pond (pond 3), we combined all taxa into fish counts and fish occurrence in subsequent analyses.

We collected a total of 324 total point counts and >55 h of video footage of fish abundance and occurrence in dense submersed aquatic vegetation habitats from July to October 2011. Our ability to obtain sufficient (i.e. ~15–20) sample numbers per sample week varied due in part to the onset of central Florida's storm season and high winds and heavy rain associated with Hurricane Irene and other large storm systems from late-August through October. The onset of the storm season may have contributed to variability in count and occurrence data due to increased observation error due to cloudy and high wind conditions. For all sampling, we asked Seaview, Inc. to disable the internal IR LEDs in the UVC to reduce backscattering on floating particulates and submersed aquatic vegetation stems. This backscattering substantially limited the field of view in initial trials, and thus, we only used the camera without the LED for all point counts in this study. We note that the automated LEDs could prove useful in other low-light or turbid systems.

The video analysis of UVC point counts were able to observe, identify, and enumerate juvenile and adult *Lepomis* spp. and juvenile and adult largemouth bass *Micropterus salmoides* in dense, surface-matted submersed aquatic vegetation habitat (Fig. 3). In total, fish were present in 179 of 324 point counts across all three ponds. We observed adult largemouth bass in 2 of 106 point counts (as largemouth bass were only in pond 3 which was sampled 106 times); juvenile largemouth bass in 9 of 106 point counts; *Lepomis* spp. in 175 of 324 point counts and unidentified fish in 4 of 324 point counts. Every largemouth bass observation also had *Lepomis* spp. co-occurring in the same locations. Further, our highest *MaxN* was two for adult largemouth bass, two for juvenile largemouth bass, and 21 for *Lepomis* spp. indicating that UVC video can be used to distinguish between species and different sizes of fish and enumerate many individuals. Table 3 shows an example video count for *Lepomis* spp. for a case where the interval would be recorded as having a $MaxN_{Lepomis}$ of two.

The UVC sampling gear functioned well in surface-matted submersed aquatic vegetation environments of high plant biomass and high habitat complexity. Surface coverage of hydrilla did not change during the study period remaining ~100% throughout the duration of the study and the surface canopy was consistently at 0.7–1.0 m in thickness. Plant biomass was high and did not significantly differ among ponds (ANOVA $P = 0.906$, degrees of freedom (d.f.) = 102, $F = 0.099$) with pond 1 mean plant biomass at 2.11 kg m^{-2} dwt (95% confidence interval (CI) 1.62–2.60 kg m^{-2} dwt), pond 2 at 2.14 kg m^{-2} (95% CI 1.55–2.74 kg m^{-2} dwt), and pond 3 at 2.26 kg m^{-2} (95% CI 1.82–2.71 kg m^{-2} dwt). Among all ponds, plant biomass significantly decreased over each month (linear GLM $P = 0.004$, d.f. = 103, $F = 8.255$) with mean plant biomass of 2.68 kg m^{-2} dwt in July, 2.64 kg m^{-2} in August, 2.00 kg m^{-2} in September, and 1.54 kg m^{-2} in October. Habitat complexity (measured with the relative stem count) sampled from underwater video differed significantly between ponds (ANOVA $P = 0.001$, d.f. = 321, $F = 9.30$) with habitat complexity highest in pond 1 (mean stem count 59.5, 95% CI 56.1–63.0), intermediate in pond 2 (51.8, 95% CI 47.8–55.9), and lowest in pond 3 (48.3, 95% CI 44.6–52.0), however complexity did not significantly change over the sampling season (linear regression $P = 0.353$, d.f. = 322, $F = 0.867$). Though the differences in habitat complexity across the ponds were significant, they did not represent a large qualitative difference as the mean stems counted (60 in pond 1, 53 in pond 2, and 48 in pond 3) all represented what we considered intermediate habitat complexity. For example, low habitat complexity occurred around 0–25 countable stems and video was not obscured by much vegetation with ~100% of the field of view observable for fish sampling. Intermediate habitat complexity occurred around 25–65 stems and particularly large patches of stems could block fish from view, but fish typically swam into an unobscured field of view at some point during their time onscreen. High habitat complexity occurred around 70 stems and greater where large patches of stems occurred more frequently creating more onscreen obstruction. As such, sampling fish in high habitat complexity required slower playback and more careful video observation.

The qualitative differences in the true population size (i.e. low, medium, and high) and population trajectories in each pond were accurately reflected in UVC fish occurrence and *MaxN* between the ponds (Table 4). Fish occurrence and *MaxN* were lowest in pond 1, intermediate in pond 2, and highest in pond 3 (Table 4) and these differences were significant (binomial regression $P < 0.0001$, d.f. = 321, $F = 85.25$; negative binomial regression $P < 0.0001$, d.f. = 321, $F = 80.54$). Fish counts, reported as *MaxN*, increased with true fish abundance among ponds, suggesting that video counts captured changes in true population sizes (Fig. 4, upper panel). In ponds with stable fish populations (pond 1 and pond 3) the *MaxN* did not significantly change through the season (negative binomial regression $P = 0.554$, d.f. = 109, $F = 0.353$; $P = 0.201$, d.f. = 104, $F = 1.653$ respectively); however *MaxN* in pond 2 increased through the season with increases in recruitment of young *Lepomis* spp. (negative binomial regression $P < 0.0001$, d.f. = 105, $F = 24.67$; Fig. 4 upper panel). Fish occurrence also increased with increases in true fish abundance. The fish occurrence in pond 1 and pond 3 did not have significant changes through the season (binomial regression $P = 0.333$, d.f. = 109, $F = 0.945$; $P = 0.766$, d.f. = 104, $F = 0.089$ respectively; Table 4), but pond 2 had significant increases (binomial regression $P < 0.0001$, d.f. = 105, $F = 29.34$; Fig. 4 lower panel). Thus, the UVC was able to detect differences in true fish abundance among the three ponds and within pond 2 as true fish abundance increased over the season.

The nonparametric bootstrap of the start and end sample weeks revealed that the ability to detect differences among fish populations depended on the population size in the ponds and

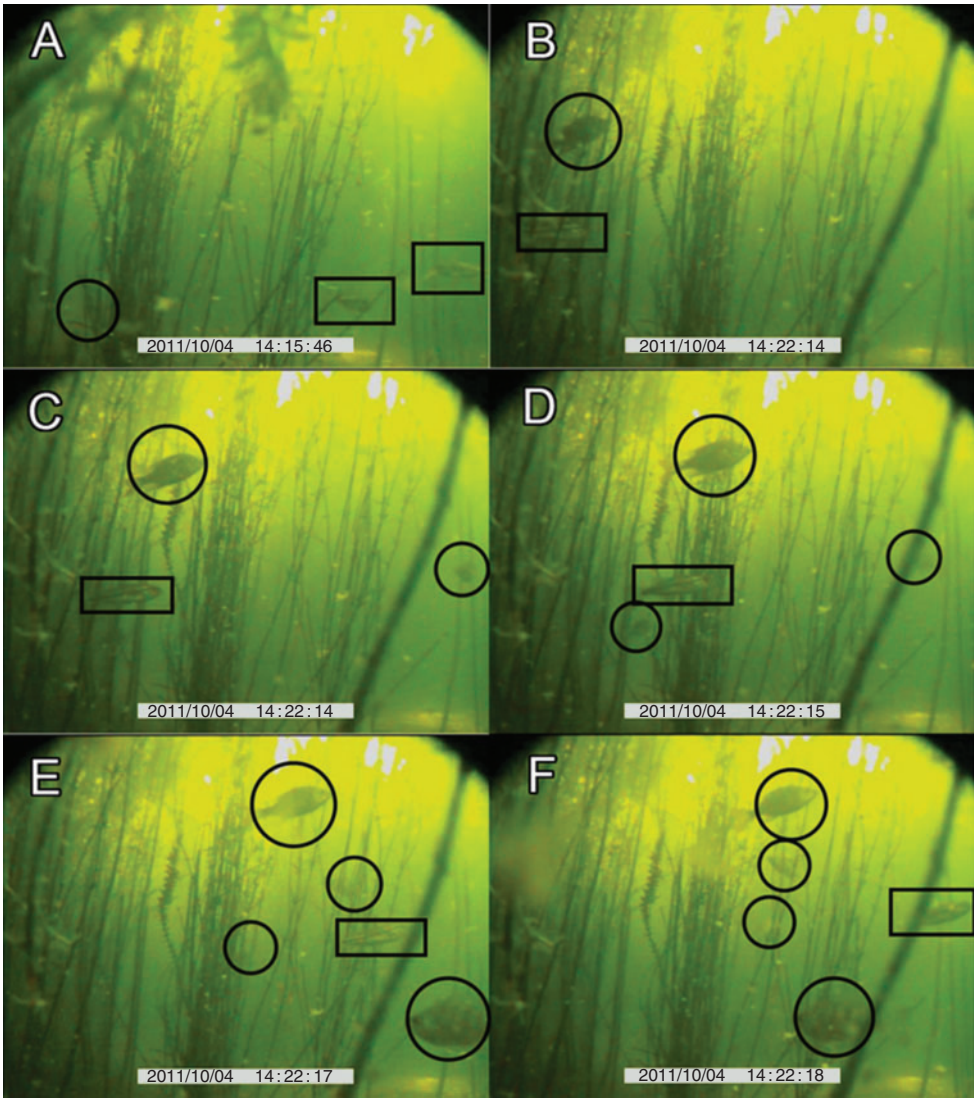

Fig. 3. Example underwater video camera point count taken underneath dense, surface-matted hydrilla highlighting the maximum number of fish (*MaxN*) for juvenile largemouth bass (*MaxN* = 2; in rectangles) and *Lepomis* individuals (*MaxN* = 4; in circles) occurring at different times. Two juvenile largemouth bass (panel A) were observed with one *Lepomis* individual in the beginning of the video. By the end of the video (panels E and F) only one juvenile largemouth bass was observed with up to four *Lepomis* spp.

Table 3. Example video count taken in October 2011 on pond 3 broken down into 30 s intervals (minutes:seconds)

The maximum number of fish (*MaxN*) is shown for each species in each time interval, the time interval where *MaxN* occurs is noted, and each fish at time of *MaxN* was classified as either adult or juvenile from morphological characteristics

	Time interval for 10 min video sample							
Fish type	4:00	4:30	5:00	5:30	6:00	Time of *MaxN*	Juvenile	Adult
Lepomis spp.	2	1	1	1	1	4:00	1	1
M. salmoides	0	2	0	0	1	4:30	2	0

Table 4. Three ponds with dense submersed aquatic vegetation with different end-of-season fish densities (fish m^{-2}) were sampled with an underwater video camera system

The mean fish occurrence (presence/absence of fish) and the mean maximum number of fish onscreen at one time (*MaxN*) were taken for all fish taxa and all sampling periods in that pond. Trend refers to how those metrics responded over the course of the sampling season according to regression analyses

Pond	True fish density	Density trend	Fish occurrence	Occurrence trend	*MaxN*	*MaxN* trend
1	0.01	Stable	0.23	Stable	0.34	Stable
2	1.23	Increased	0.50	Increased	1.63	Increased
3	3.70	Stable	0.94	Stable	5.03	Stable

the number of UVC samples per week. In ponds with high fish densities (i.e. the start/end of pond 3 and end of pond 2), at least 20 samples per week were needed to detect a 3.0× population difference with Student's *t*-test (Table 5). With larger true population differences (e.g. 40× or more) relatively fewer samples were needed to detect that population difference. In ponds with low fish densities (i.e. the increased frequency of zero-counts in pond 1 and the start of pond 2), Student's *t*-test could not detect a significant difference between populations that differed by 2.0–3.5× even by increasing the samples per week to 30 (Table 5). The *MaxN* counts that sampled the starting, unknown fish population size in pond 3 were not significantly different to the *MaxN* counts of the known population in week 13 across all sample intensity scenarios (Table 5), which supported our assumption that pond 3 was at a consistently high fish density throughout the sample season.

The nonparametric bootstrap suggested that 20 UVC samples per week was the most optimal sampling strategy given our observed count distributions. Using 20 samples per week in the bootstrap simulation, the estimates of relative population size from UVC *MaxN* counts appeared to scale linearly with true population estimates (Fig. 5). The 95% confidence intervals for 20 samples per week for the UVC estimates of relative abundance for low (21–150 fish per pond), intermediate (5000 fish per pond), and high (>15 000 fish per pond) population sizes did not overlap (Fig. 5). The 3.0× difference in the end of season fish populations between pond 2 and pond 3 was reflected in a 2.7× difference in the mean *MaxN* between pond 2 (mean

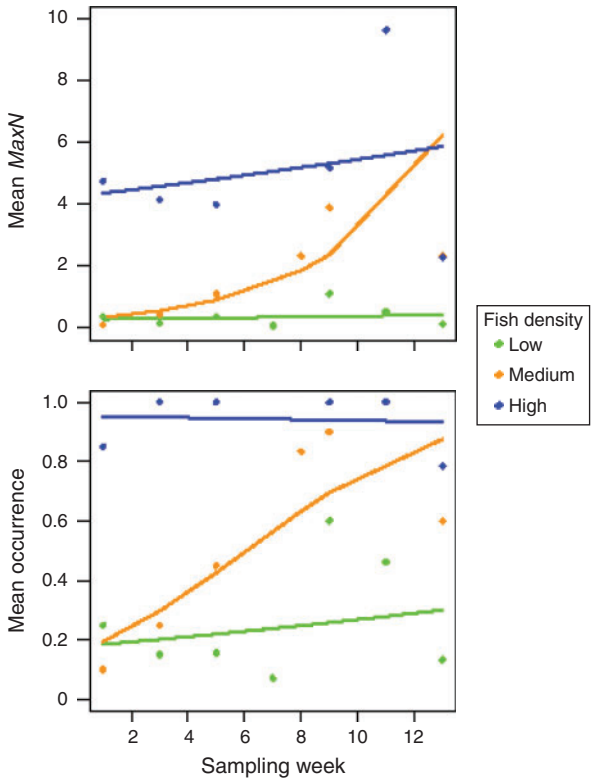

Fig. 4. The observed (points) and predicted (lines) mean maximum number of fish onscreen (*MaxN*; upper panel) and mean fish occurrence (lower panel) for all fish sampled in dense submersed aquatic vegetation with underwater video in each pond per sampling period. Note – the week 13 mean *MaxN* for medium and high fish density overlap.

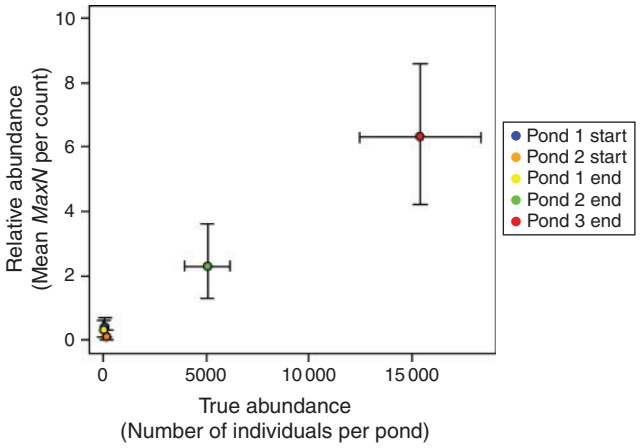

Fig. 5. Estimated relative fish abundance (quantified from underwater video) regressed onto estimated true fish abundance (quantified from maximum likelihood estimates from pond draining and quadrat collections with associated 95% confidence intervals). The mean and 95% confidence intervals for estimated fish abundance were derived from nonparametric bootstrapping (10 000 iterations at 20 resamples) of video count data from the start and end (weeks 11 and 13) sample weeks.

Table 5. The percentage of significant ($\alpha = 0.05$) one-way or two-way (depending on the ponds compared) Student's *t*-test for 10 000 iterations from a nonparametric bootstrap testing for differences in video count data taken at the start (week 1) and end (weeks 11 and 13) sample weeks at different sampling intensities

		Sample number scenarios				
Ponds compared	True population differences	10	15	20	25	30
Pond 3 end > Pond 2 end	3.0×	70	87	95	98	99
Pond 3 end ≠ Pond 3 start	'Carrying capacity' assumption	10	14	18	22	25
Pond 3 end > Pond 1 end	714×	100	100	100	100	100
Pond 2 start < Pond 2 end	40×	100	100	100	100	100
Pond 2 end > Pond 1 end	285×	76	94	99	100	100
Pond 1 start > Pond 1 end	3.5× (Low density)	4	5	5	6	6
Pond 1 start < Pond 2 start	2× (Low density)	2	1	0	0	0

MaxN of 2.3, 95% CI 1.3–3.6) and pond 3 (mean *MaxN* 6.3, 95% CI 4.2–8.6). As well, the 33× difference between the start of pond 2 and the end of pond 2 was reflected with a 23× difference in UVC estimates (start pond 2 mean *MaxN* 0.1, 95% CI 0.0–0.3; end pond 2 mean *MaxN* 2.3, 95% CI 1.1–3.8) and the 100× difference between the start of pond 2 and the end of pond 3 was reflected with a 63× difference in mean *MaxN*. However, *MaxN* counts failed to quantify some differences among true fish populations at low densities. The *MaxN* counts from UVC estimated that pond 1 had four times more fish (mean *MaxN* 0.4, 95% CI 0.1–0.7) than pond 2 (mean *MaxN* 0.1, 95% CI 0.0–0.3) when pond 2 actually had twice as many fish as pond 1 at this time. The 230× difference between the end populations of pond 1 and pond 2 were shown to have only a 7.3× difference in UVC counts (end pond 1 mean *MaxN* 0.3, 95% CI 0.1–0.6). With at least 20 samples per week (which was not always obtained in this study due to anomalous weather) the UVC was a suitable method for estimating and quantifying relative abundance differences in some fish populations in dense submersed aquatic vegetation environments, but using *MaxN* as a relative abundance estimator did not always scale proportional to true differences, particularly at low fish abundance.

Fish did not appear to respond strongly to the boat-based UVC sampling. Observations of identifiable individuals showed that some individuals investigated the camera and many individuals stayed within the field-of-view for several time intervals. Most individuals swam at a constant pace in and out of view with no noticeable behaviour attracting towards or repelling from the camera and many natural fish behaviours were observed, such as grazing, conspecific interactions, and feeding on macroinvertebrates. During one instance in July 2011, two adult bass were recorded with 10–17 *Lepomis* spp. for upwards of 7 consecutive minutes despite slightly windy conditions on the surface affecting the boat and camera positioning. In this case, individual *Lepomis* spp. still appeared to respond more strongly to the presence of the bass by avoiding the immediate areas of the two bass, rather than responding to the presence of the UVC. Typical video footage recorded immediately after the camera was positioned (i.e. right after the camera descended) showed many fish with a constant swimming speed in the field-of-view despite the camera having just arrived to the sample area, indicating *Lepomis* spp. did not strongly avoid the camera despite its novelty. During sharp camera movements due to changes in surface conditions, fish often swam away but not out of view and within 15–30 s most fish resumed a constant swimming motion and appeared to resume natural behaviours.

Table 6. Evaluation of two linear regression models testing whether fish counts changed through the duration of video sampling with the significance of the regression model (*P*-values) and ranked with Akaike's Information Criterion (AIC) model selection

If ΔAIC < 2.0, models are considered equivalent and the most parsimonious model is selected

Model	Intercept	Slope	*P*-value	Parameters	AIC	ΔAIC
Time	0.736	−0.013	0.0481	2	26 026.28	0
Null	0.673	–	<0.0001	1	26 028.18	1.9

The number of individuals counted did not significantly change over recording-time during the 10-min UVC point counts. Both the tested regression model (that fish counts changed through the 10-min duration) and the null model significantly predicted fish counts (linear regression $P = 0.048$, d.f. = 6873, $F = 3.906$, null regression $P < 0.0001$ respectively; Table 6). The slope of the tested model was very low (−0.013) predicting only a small decrease in fish abundance from the video start compared with the video end. The two models ΔAIC scores were less than 2.0 indicating that the models were equivalent in explaining the variation in fish count data over video duration (Table 6). This suggests that the time of arrival for fish was not influenced by the presence of UVC equipment and that UVC may not repel or attract fish to the sample location.

Discussion

Underwater video cameras provided an innovative, non-lethal technique to sample fish communities, estimate relative abundance, and track population changes in dense submersed aquatic vegetation environments. This is the first study of its kind to use UVC to quantify spatio-temporal trends in relative fish abundance within dense freshwater submersed aquatic vegetation. Though the success of using *MaxN* from UVC to quantify differences among each pond's fish population varied in low fish densities, the use of *MaxN* detected qualitative differences among the different populations and their population changes over the season. An advantage of UVC to other sampling techniques in submersed aquatic vegetation, such as electrofishing, is that it allows for direct observation of fish in structurally complex habitat, which addresses concerns listed by Dibble *et al.* (1996) regarding the lack of micro-scale inferences on fish–plant interactions. Furthermore, UVC can provide several different metrics at the population (e.g. presence-absence, abundance, species diversity) and behavioural level (e.g. co-occurrence, movement, foraging) and provide permanently captured data that can be reviewed repeatedly for accuracy. The method also captures charismatic images for public/outreach media and communication.

The UVC worked well as a tool to accurately and precisely detect relative population changes at fish densities we would expect to encounter in natural systems. The total fish biomass where UVC was most effective (i.e. in pond 2 which held 89–135 kg ha^{-1} and in pond 3 which held 263–385 kg ha^{-1}) was consistent with total fish biomass across dozens of Florida lakes. In comparison, total fish biomass measures from several studies on more than 60 Florida lakes indicated a typical total fish biomass that ranges from 165–315 kg ha^{-1}, with some estimates as high as 615 kg ha^{-1} (Barnett and Schneider 1974; Kautz 1980; Chick and McIvor 1994; Bachmann *et al.* 1996). The high density pond in this study (37 000 fish ha^{-1} or 333 kg ha^{-1}) was assumed to be likely near the maximum capacity for the experimental ponds given the comparison of this study's fish densities with other Florida lakes and the multiple years allowed for the population to grow. This indicated that the use of *MaxN* video analysis of UVC data performed well when assessing fish populations that were within the bounds or near the capacity of what would be natural and could detect and quantify population changes at such fish densities.

The loss of adult fish in pond 1 (72% of stocked *Lepomis* spp.) and in pond 2 (54%) over four months were around the range for natural mortality for *Lepomis* spp. in central Florida (0.50–0.55 according to Crawford and Allen 2006, 0.43–0.65 in Parsons and Reed 1998). Further, the physical recovery of 21 adult fish in pond 1 represented the minimum estimate of adult population size at the time of the pond draining as other adult fish may have remained alive but unsampled with the nets and quadrats. We considered the changes in pond 1 fish density, from 185.2 to 51.9 fish ha^{-1} (biomass changes of 1.67 to 0.47 kg ha^{-1}) to be a small real difference in fish densities that inflated zero-counts. Further, these fish densities are uncommonly low for Florida lakes, where fish densities range from 30 000 to >1 000 000 fish ha^{-1} (~300 kg ha^{-1}; Chick and McIvor 1994; Bachmann *et al.* 1996). Given limited number of samples per week at low fish densities, we expected zero-inflation to increase variability in the count data over any sampling period. The key finding at this low population-level is that the population did not have recruitment and represented a very low and either stable or slightly declining population over four months, and this relative stability was detected by fish occurrence and *MaxN* counts. The occurrence data in pond 1 had high variation but regression models were not significant for changes over time and increased samples per week would likely reduce this variability. The nonparametric bootstrap also suggests that the *MaxN* was useful for estimating population size at these low populations with enough samples, but UVC could not effectively distinguish between populations of 370.4, 185.2, and 51.9 fish ha^{-1} due in part to zero-inflated counts. Quantifying the specific magnitude differences between low fish densities and high fish densities depended on the extent of outlier samples from low density ponds. This discrepancy is likely due to increased uncertainty from zero-inflated counts influenced by low population densities. This limits the utility of UVC to assessing populations with Poisson or negative-binomially distributed frequencies, or to create novel UVC deployments or statistical analyses for assessing zero-inflated count data of more cryptic fish species or for fish at very low population sizes.

Fish behaviour can be altered by the presence of sampling equipment (reviewed in Stoner *et al.* 2008). The number of fish counted was not influenced by the time interval within each point count, which indicates that sampling duration can be modified to improve efficiency. Shorter duration of point counts would allow for larger sample sizes, and thus, improved statistical power and efficiency. Fish arrival rates and time of *MaxN* can be different among fish species and can be a predictor variable in fish density estimates influencing the duration of sampling events for many fishes, particularly slow-moving or cryptic species (Priede and Merrett 1998; Cappo *et al.* 2004). Ebner and Morgan (2013) found most freshwater fish species in Australian waterholes were detected within 10 min of underwater camera deployments while Ellender *et al.* (2012) found that two imperilled freshwater fish in African streams were observed within 15–22 min. This supports our assumption that any sampling or behavioural biases within *Lepomis* spp. could be detected within 10 min of sampling. Because fish were not significantly affected by the presence of UVC in this study, future work with this sampling design can assume that differences in time of arrival or time of *MaxN* are naturally occurring and not a result of fish response to the sample gear.

The habitat conditions in each pond varied due, in part, to differences in habitat complexity measured from UVC and that complexity was decoupled from plant biomass. Though biomass changed seasonally, possibly due to changes in the thickness and density of the surface canopy, the hydrilla stems that provided structure at the depths sampled for fish did not appear to change over the duration of the study. Our measure for habitat complexity, the relative stem count sampled from the camera's perspective, quantified the number of stems onscreen that might partially influence the visible range of the camera. Since habitat complexity did not change over the sampling season, our study should not have a bias on fish presence/abundance in the UVC metrics from the beginning compared with the end of the season. Additionally, any bias caused from habitat complexity influencing detection probability was likely biased in the same direction and magnitude since complexity was qualitatively similar between the ponds. We applied no correction in detection probability to adjust *MaxN* counts dependent on habitat conditions and were still able to effectively estimate and track fish populations in the ponds. Future adaptations of the UVC technique may need to apply a detection probability correction to count data based on habitat complexity as recommended in Gu and Swihart (2004) and Williams *et al.* (2002). Our study indicated that such a correction is not strictly needed, as the *MaxN* derived from UVC appeared to be an unbiased estimator of relative fish abundance.

Video data of fish provide several different indicators of presence and/or abundance, including: (1) presence–absence or occurrence of fish; (2) maximum count or the total number of fish of each taxon observed over an entire video sample (Chidami *et al.* 2007); (3) maximum number of fish of each taxon onscreen at one time (*MaxN* in Cappo *et al.* 2004; *mincount* in Gledhill *et al.* 2005); and (4) a mean count (the average number of fish present in the video frame over the course of a video sample; a new video metric *meancount* provided in Conn 2011). Occurrence, *MaxN* and *meancount* have advantages over maximum fish encountered to track true fish abundance by avoiding the potential of double counting individuals. Simulations by Conn (2011) have found that *meancount* may provide an unbiased index of true abundance, while *MaxN* may be slightly underestimating trends in true abundance but provides less variability. There is no consensus on using *MaxN* or *meancount* to analyse video, although we recommend that future experimental work in dense submersed aquatic vegetation should evaluate the use of both as *meancount* could aid in quantifying magnitude differences between low and high fish densities by reducing the weight of outlier samples in low densities. In our study, *MaxN* appeared to scale linearly with population size and provided an unbiased estimator of abundance, but we lacked replication for fish density treatments. In freshwater littoral ecosystems where species can be particularly cryptic, *MaxN* was a useful and easy metric to obtain for tracking changes in relative abundance across a wide gradient of population levels, but the success of this approach varied at lower population levels. Fish occurrence and *MaxN* metrics accurately qualified all low density populations as low in relative abundance. The application of underwater camera techniques to

sample uncommonly low fish densities (51.9 fish ha^{-1} or 0.47 kg ha^{-1}) in dense submersed vegetation can accurately assess and qualify such populations as low according to estimates from Bachmann *et al.* (1996). This makes these techniques useful in identifying areas and populations of concern that were, previously, often poorly assessed.

Our study focussed on quantifying population trends of relatively common and important recreational fishes in dense, invasive submersed aquatic vegetation habitats. We decided on using non-baited cameras over baited cameras due to how common the targeted fishes were in Florida lakes and the complications in calculating 'bait plumes' for evaluating the total area sampled in baited camera surveys (reviewed in Cappo *et al.* 2007). We encountered design problems and limitations for using UVC in dense, surface-matted submersed aquatic vegetation and decided upon using these particular cameras mounted from the boat. In preliminary trials, we attempted to use transect-surveys, fixed-placement UVC (i.e. not boat mounted), and a cone over the UVC to aid in canopy penetration, all of which snagged on plants and obstructed view. For broader application in vegetated lentic environments, modifications to the boat-based deployment are recommended to reduce disturbances to fishes and potentially increase species-diversity sampled with underwater video. Though this study's boat-based approach indicated minimal observer effects on *Lepomis* spp. and largemouth bass, we recognise that this may not apply to all fish species. For example, prey fishes may be much more risk-averse to the novel stimuli of a boat and camera and avoid the sample site altogether. In preliminary work sampling the fish community inhabiting hydrilla on Lake Tohopekaliga, Florida, we slightly modified UVC deployment using an airboat (commonly used boats for working in aquatic vegetation) and tethered the camera to a buoy with 10–15 m of slack cable and then tethered the buoy to the airboat. The airboat appeared to minimally disturb the vegetation and any disturbance within the airboat (e.g. windy conditions moving the boat) had no impact on the camera due to the slack cable. In this approach, boat-based observer effects are minimised while count replicates are still easily obtained. Preliminary results from the Lake Tohopekaliga work in summer 2012 found the airboat-based UVC approach recorded fish across a range of taxa and trophic levels (e.g. largemouth bass, sunfishes, bluefin killifish *Lucanea goodei*, lake chubsucker (*Erimyzon sucetta*) and chain pickerel (*Esox niger*); K. Wilson, unpubl. data). We recommend the boat-based UVC point counts to sample fish in dense vegetation with some consideration on the specific boat and deployment tactic to minimise observer effects.

The sampling protocol used in this study design followed point-transect sampling typical in wildlife studies (Robbins *et al.* 1986; Buckland 2006), more than the multi-camera deployments used in baited remote underwater video surveys (e.g. Ebner and Morgan 2013). Each point count was an instantaneous count (e.g. *MaxN*), with repeated-measures from a single observer. Due to the video analysis protocol, double-counting of individuals was not possible within a point count, but the observer moving across the pond could lead to double counting individuals among point counts, particularly of highly mobile fish. This concern is best addressed with statistical analyses that can account for spatial-dependence. We recommend that UVC approaches be considered as suitable tools (over electrofishing in some instances) in creating cost-effective fisheries monitoring programs with permanently stored video data for repeated long-term analyses. In applying boat-based UVC methodology to broader fisheries management contexts, statistical power and spatial-dependence are likely two of the most critical considerations to designing effective monitoring programs. Such a program can follow advice from the North American Breeding Bird Survey on long-term monitoring of populations with point-transect data (Robbins *et al.* 1986; Mathewson *et al.* 2012). We advise that at least 20 point counts should be taken to quantify population trends at a variety of sample locations within a lake and across several lakes to make long-term comparisons of fish populations. Spatial-dependence can be accounted for in a variety of manners. For example, analysing count data with a negative-binomial distribution or using generalised least-squares for fitting regression models can help account for spatial dependence. The location of each point count should be sequential or have spatially explicit coordinates measured. Spatial-autocorrelation can negate inferences made in count data, but these factors can be controlled with recent statistical techniques (e.g. 'spdep' in Program R; Bivand *et al.* 2013) by accounting for spatial-dependence with an autocovariate parameter applied in regression analyses. Well developed monitoring programs could be relatively affordable to implement as cameras are cost-effective sources of information. Such a program would create a long-term dataset across regions that can be evaluated for changes in fish populations over time and space due to habitat and climate changes (Link and Sauer 1998; Mathewson *et al.* 2012).

The specific utility and deployment of UVC for monitoring programs will depend on specific questions that merit addressing via monitoring. Evaluating fish community-level responses to invasive plant habitats, and subsequent aquatic plant management, may require longer video sets (from 20 min to several hours) to quantify species richness, a commonly utilised community-level metric (e.g. Ebner and Morgan 2013). Since fish behaviour can differ drastically across taxa, observing a wide-variety of fish species in a given location, including cryptic and sedentary fishes, may take upwards of several hours. To evaluate specific taxa, including endangered and threatened species, a program that takes into consideration the specific habitats and behaviours of those fish will need to be designed before applying UVC methodology (e.g. Ellender *et al.* 2012). Any application of our boat-based UVC methodology to other environments should consider the influence of depth of deployment and habitat complexity on fish detection probability. Ebner and Morgan (2013) found that the camera deployment in shallow, vegetated environments observed different freshwater fishes than cameras deployed in deeper waters, and the influence of baited cameras slightly increased the detection of some species. Future application may also improve by evaluating the time of arrival for a variety of fishes to establish effective deployment times and quantifying behaviours observed among individual fishes to quantify habitat use (e.g. foraging, refuge, or nursery habitats) in these newly sampled environments. We caution against relying solely on one single type of method or on one single deployment tactic to fully sample a fish community, but, as Ebner and Morgan (2013) point out, some tactics will provide the highest efficiency in sampling the most fishes.

We showed that the UVC can be useful for assessing fish abundance in dense submersed aquatic vegetation habitats, which have traditionally been difficult to sample with standard gears such as electrofishing or fyke nets (Bayley and Austen 2002). The UVC effectively measured differences in fish abundance among the ponds and detected most population changes. Resource managers could use these methods to evaluate fish community responses to changes in habitat (e.g. sample for littoral fish community's response before/during/after aquatic plant removals) or to detect differences in fish abundance among lakes with different habitat characteristics. For example, freshwater fish frequently inhabit areas of low dissolved oxygen or impacted water quality, and a sampling protocol that quantifies fish responses, taken from UVC metrics, and habitat/water quality metrics might be superior for time course sampling of fish communities in hypoxic environments compared with traditional methods. An additional utility in using UVC point counts is in the emerging field of occupancy modelling (Williams *et al.* 2002). The UVC technique offers an ability to conduct repeated-measures at a sample location with little disturbance that could be used to estimate habitat-specific detection probabilities and gain inferences on site occupancy and animal abundance (MacKenzie *et al.* 2002; Williams *et al.* 2002; Royle and Nichols 2003). Occupancy modelling offers new opportunities to more effectively quantify fish populations, habitat associations and population responses to changes in habitat quality and quantity in an environment that has been historically understudied due to impaired statistical inference.

The UVC could also be used to evaluate a variety of ecological and management objectives in freshwater ecosystems. Trophic interactions in the aquatic food web could be evaluated by sampling for multi-species occurrences across benthic, pelagic, and littoral fish communities (e.g. Ebner *et al.* 2009). Video cameras could track spatio-temporal patterns in fish behaviour and/or habitat utilisation in a variety of habitats. Sampling designs could be focussed on areas, depths and seasons to evaluate a particular species of concern for conservation (e.g. Ellender *et al.* 2012). The use of the UVC requires water to have adequate transparency, which limits its utility in highly turbid systems. However, many other systems would allow use of UVC to measure fish occurrence and relative abundance, and these methods could be explored relative to traditional sampling gears in a range of freshwater systems with high levels of aquatic vegetation that are actively managed for invasive plant species or flood-control (e.g. rivers, lakes, and ponds). We encourage future efforts to continue to explore the use UVC to assess fish communities in freshwater systems.

Acknowledgements

The authors thank the Florida Fish and Wildlife Conservation Commission Invasive Plant Management Section for financial support and the United States Geological Survey South-eastern Ecological Science Center for logistics and access to the experimental ponds. They would also like to thank the University of Florida's Dan Gwinn, Ed Camp, and Jeremy Slade for advice during the development of this manuscript, as well as Erin Bradshaw, Nicholas Cole, Zack Slagle, Simone Nageon de Lestang, and Antonio Malouf for field collections and video analysis. We thank the Associate Editor, Russell Death, as well as Brendan Ebner and three anonymous reviewers for the feedback to improve the manuscript.

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