

Countering low visibility in video survey of an estuarine fish assemblage

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Abstract. Sampling fish communities in tropical estuaries is inherently challenging due to poor visibility and the potential presence of dangerous fauna. We present two strategies for improving the identification of fishes in a turbid tropical estuary using video. The first was to attract species close to the camera by using two different bait types compared with no bait, and the second involved manipulating footage in the postfilming phase. No significant difference was found in the species richness recorded among camera bait treatments (thawed Australian sardines, canned sardines and unbaited), although baited cameras did detect 13 taxa not observed on the unbaited cameras. Three different image restoration algorithms (histogram equalisation, white balance and contrast-limited adaptive histogram equalisation) were compared in processing 22 instances where fish could not be confidently identified to species or genus level. Of these processed clips, five were able to be identified to species level by a panel of four coauthors. Further, two of the three algorithms yielded higher average confidence values for identification at the order, family, genus and species level than when the unprocessed footage was viewed. Image restoration algorithms can partly compensate for a reduction in image quality resulting from turbidity, addressing a key challenge for video-based sampling in estuaries.

Additional keywords: bait, image restoration, tropics, turbidity.

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Introduction

Remotely deployed underwater video has been used extensively in marine systems to survey fish assemblages (e.g. [Priede et al. 1990](#); [Cappo et al. 2004](#); [Langlois et al. 2006](#)). However, the emergence of the technique is relatively recent in freshwater and estuarine systems ([Meynecke et al. 2008](#); [Ebner et al. 2014](#)). There have been several useful applications to threatened aquatic fauna, with the non-destructive aspect of this approach suited to repeat monitoring of such species ([Butler and Rowland 2009](#); [Ebner et al. 2009, 2014](#); [Ellender et al. 2012](#)). One key advantage of remotely deployed underwater video is that it can be used to safely observe and identify aquatic species during ecological field-based research in dangerous or difficult-to-access locations ([Gomelyuk 2009](#); [Ebner et al. 2014](#)). In the

tropical Pacific, estuaries are occupied by dangerous species including box jellyfish and large predators such as crocodiles and sharks, rendering direct observation by snorkelling and scuba diving unsafe. In these circumstances, and even where these threats are absent, boat- or shore-based deployment of passive or active survey techniques including remotely deployed underwater video, may be favourable. However, water clarity can be periodically or continuously compromised in estuaries ([Cyrus and Blaber 1992](#)), presenting challenges for visual survey methods and specifically the detection and identification of taxa by observation. Furthermore, the video-based identification of a species is influenced by several factors including the size and contrast of the focal organism (and specifically its defining features), the mobility of the organism, the distance between that

organism and the camera, the resolution of the camera, light availability and researcher expertise (Carleton and Done 1995; Ninio *et al.* 2003; Mallet and Pelletier 2014).

Remote underwater video has been used to survey mobile fauna in several low-visibility aquatic habitats including estuaries (Meynecke *et al.* 2008; Becker *et al.* 2010; Fulton *et al.* 2012; Unsworth *et al.* 2014). This research has included applications of baited and unbaited cameras, with baits sometimes drawing target species close enough to the camera to enable positive identification under low-visibility conditions (e.g. Fulton *et al.* 2012). Varying responses to bait type have been reported in previous studies, with some finding only subtle differences between baited and unbaited treatments (e.g. Ebner and Morgan 2013; Cousins *et al.* 2017), while Harvey *et al.* (2007) reported that the likelihood of detecting species on baited or unbaited cameras varied on the basis of feeding guilds. Studies using baited cameras in remote locations also reported that canned or dry foods offer substantial logistical benefits over frozen alternatives, particularly on field expeditions in the tropics or in temperate areas during warm weather (Ebner *et al.* 2014). Optimal camera deployment duration is another issue that has been explored by previous studies (e.g. Ebner and Morgan 2013; Harasti *et al.* 2015). There are several ways to measure the most appropriate deployment duration and this may depend on the objectives of the study. For estimating species richness, there are generally diminishing returns after a certain deployment length, therefore determining optimum deployment durations for a given setting is an important factor in survey design.

Infrared and natural-spectrum lighting have been used to illuminate fauna in darkened scenarios in aquatic ecosystems (Priode *et al.* 1990; Schmitt and Holbrook 1999; Aguzzi *et al.* 2009; Ebner *et al.* 2014). In some cases, recognition of species has been automated or partly automated to overcome manual labour requirements of the video-processing phase in projects (Aguzzi *et al.* 2009; Salman *et al.* 2016). However, there has been limited exploration of using image enhancement in the postprocessing of video in aquatic ecological research.

Light travelling through water suffers scattering and absorption that can affect the quality of underwater images, generating contrast loss and colour shift. The magnitude of distortion on an image is linked to the distance between the scene and the camera lens, with objects further away being more distorted. Several algorithms have been proposed for underwater image restoration (Carlevaris-Bianco *et al.* 2010; Chiang and Chen 2012; Codevilla *et al.* 2014; Drews *et al.* 2015). He *et al.* (2011) proposed an approach for weather-degraded images, and this is now one of the most cited papers in the field of underwater imagery due to the similarities between weather-degraded and underwater images. He *et al.* (2011) proposed the Dark Channel Prior (DCP), a statistical method based on the observation that natural images exhibited a mostly dark intensity in a square patch in at least one colour channel of the images. In the aquatic medium, the main assumption stated by He *et al.* (2011) is plausible for which at least one colour channel has some pixels whose intensity are close to zero. These low-intensity pixels occur because of shadows, objects or surfaces where at least one colour channel has low intensity (e.g. corresponding with certain fish, algae or corals). One assumption made by He *et al.* (2011), that any effects produced by the medium are

independent of wavelength, does not hold true for typical underwater conditions, where there are high absorption rates in the red channel. Thus, Drews *et al.* (2013) proposed use of the Underwater Dark Channel Prior (UDCP), which assumes that images of natural settings contain pixels representing very low intensity in the green or blue channels.

In the current study, baited remote underwater video stations (BRUVSs) and unbaited remote underwater video stations (UBRUVSs) were used to rapidly survey mobile fish species in a remote estuary on the eastern coast of Cape York Peninsula in northern Australia. The three primary aims were: (1) to trial postprocessing of video footage using UDCP algorithms to improve researcher confidence in fish identification; (2) to determine whether bait increased the number of species detected on video cameras, and specifically to compare a frozen and a canned bait due to foreseen practical benefits; and (3) to investigate optimum deployment duration for cameras in the context of rapid assessments (i.e. ≤ 1 h set time).

Methods

The survey was conducted in the upper reaches of the Nesbit River estuary (~ 5 km from the river mouth) on the eastern side of Cape York Peninsula, Queensland, Australia ($13^{\circ}32'53''$ S, $143^{\circ}32'38''$ E). The catchment drains ~ 700 km² and is dominated by open *Melaleuca* forest with riparian areas predominantly rainforest. The climate of the area is governed by monsoonal weather patterns and receives an average annual rainfall of 1350 mm (Silver Plains Station, station no. 028017: Bureau of Meteorology 2016a).

In November 2014 (dry season), 12 BRUVSs and six UBRUVSs were deployed along ~ 1 km of the upper estuary of the Nesbit River. Each camera was randomly allocated to one of three bait treatments: unbaited, baited with canned sardines (Clupeidae), and baited with thawed Australian sardines (*Sardinops sagax* (Jenyns, 1842)). Each BRUVS/UBRUVS consisted of a T-shaped metal base-plate (450 mm $\times$ 150 mm) to which a single 1-kg lead dive weight and a GoPro camera (Hero2; www.gopro.com) in a dive housing were fixed at one end and two bamboo canes fixed at the other. Each GoPro camera was set to record at 960 p and 30 fps using the wide-angle setting. For baited treatments, two modifications to the setup were used. Cameras baited with sardines used two stacked tins (125 g of sardines in spring water) fixed by cable ties to the end of the canes at ~ 50 cm from the camera. The lid was removed from the upward-facing tin and the bottom tin was pierced with several holes. For cameras baited with thawed Australian sardines, a fine-mesh (5 mm $\times$ 5 mm) bag (to prevent bait depletion) and a coarse-mesh (80 mm $\times$ 80 mm) bag were used to secure three prethawed sardines at 50 cm from the camera (Fig. 1a). A short length of wire and a rope with a small float were affixed to the base plate and used to deploy and retrieve the setup from a boat. We ensured that cameras were facing away from the sun for all shallow-water deployments to minimise sunlight interference.

Camera deployments were randomised across sites with a range of depths (~ 0.3 – 2.5 m), substrata and proximity to structure and were spaced ~ 50 m apart. The cameras were deployed in two sets of nine during the course of a single day, with all cameras in each batch deployed almost simultaneously

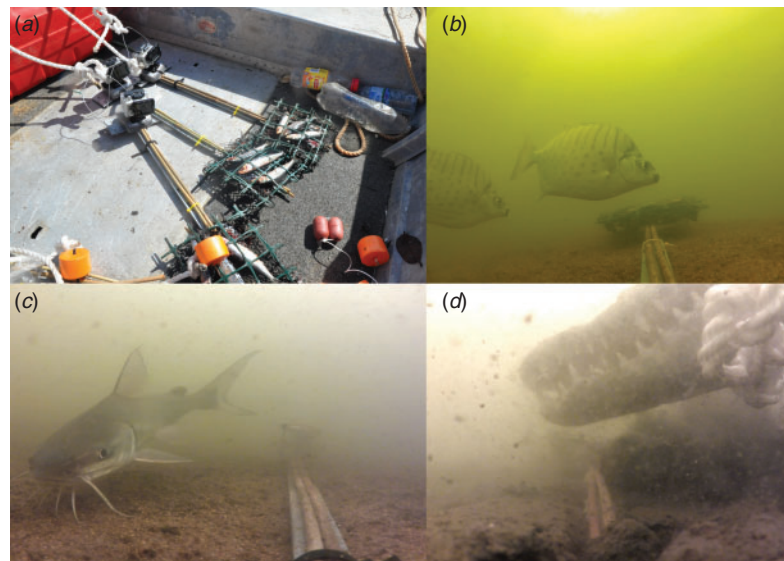


Fig. 1. Photographs from the study: (a) cameras baited with pilchards ready for deployment, and screenshots taken from video footage of (b) *Selenotoca multifasciata*, (c) *Ariidae* sp., and (d) *Crocodylus porosus*.

(within 10 min). Each camera was deployed for at least 1 h. Horizontal visibility through the water column was estimated to be in the order of 1–3 m as a function of tide and microhabitat. As this study was conducted towards the end of the dry season, river discharge was negligible (November 2014 rainfall ≈ 1.4 mm; Coen Airport Evap., station no. 027006: Bureau of Meteorology 2016b) and any water movement was largely tidal.

Video footage collected during deployments was viewed and analysed in VLC Media Player 2.0.4 (VideoLAN, Paris, France). Footage from each camera was analysed commencing 1 min after field deployment to the benthos to minimise the influence of the boat on animal behaviour. Footage was viewed by the senior author in real time for the first 10 min after deployment and then viewed in fast forward or real time depending on ease of viewing (i.e. diversity of species in view, clarity of water and glare) for the remaining 50 min. For all identifiable species, MaxN (maximum number of a species in view at one time, following Willis and Babcock (2000) and Cappo *et al.* (2004)) and time of arrival were recorded. Instances where animals could not be confidently identified to genus or species level were noted and later treated using image-restoration techniques and subsequently manually scored by the authors. Two of the deployed cameras ceased recording before being retrieved and only recorded 31 min (thawed Australian sardines) and 38 min (canned sardines), respectively; one due to burial in soft sediment as a result of crocodile activity and the other due to a technical malfunction. These cameras were included in the analysis as the species accumulation curves showed that 80% of species were detected within the first 30 min. Two cameras, one unbaited and one baited with canned sardines, did not detect any fish during their deployments, although the latter recorded only 38 min of footage.

Twenty-two instances where fish could not be confidently identified to species or genus level were tested; 10 were on cameras baited with canned sardines, four were on cameras

baited with thawed Australian sardines and eight were on unbaited cameras. Each instance was extracted from the original deployment footage as a 5–15-s video clip. Each video clip was processed in a two-step process. The first step was to apply UDCP to improve overall image quality (colour fidelity and contrast). The clips were then processed using three postprocessing techniques: histogram equalisation (Szeliski 2010), white balance (Szeliski 2010), and contrast-limited adaptive histogram equalisation (CLAHE) (Zuiderveld 1994).

Each of the four expert reviewers were randomly allocated one of the four treatments of each of the 22 clips, such that each of the four treatments of each individual clip was examined by a different reviewer. This ensured that assessment of the four postprocessing treatments was independent. Reviewers were instructed to classify the target fish in each video clip to order, family, genus, and species level and provide a confidence value for each classification. All instances where any one of the researchers could not identify the target to genus or species level were noted as unidentified species with the exception of *Ariidae* species, which could be confidently classified only to family level due to taxonomic difficulties with this group (P. Kailola, pers. comm.).

Data analyses

The relationship between camera bait treatments and number of taxa detected was explored using a one-way ANOVA, with 95% confidence intervals calculated using a t-distribution performed in R 3.1.3 (R Core Team; www.r-project.org). Only individual fish that could be identified to genus or species were included in analyses, with the exception of species of *Ariidae* which could be identified only to family level but were included in analyses. Species-accumulation curves were also generated for each camera bait treatment to gain a preliminary indication of optimum soak times for future applications of BRUVSs in tropical estuaries.

To investigate the effect of each algorithm on confidence levels associated with the identification of fish by coauthors, we used one-way ANOVAs to compare mean confidence values associated with each treatment for each taxonomic level. *Post hoc* two-sample *t*-tests were used to compare differences between treatments within each taxonomic level. The α level (0.05) was adjusted to 0.008 using a Bonferroni correction. To further evaluate the effect of each treatment on taxonomic resolution achieved by coauthors reviewing the clips, we compared the mean proportion of identifications made with $\geq 90\%$ confidence for each taxonomic level.

To visually analyse the species composition of the video samples, MDS ordination was carried out on both taxonomic occurrence data and trophic functional group occurrence data. For both ordinations, a two-dimensional solution was produced by calculating Jaccard's coefficient similarity matrices based on binary presence/absence data. For the taxonomic ordination, rare species (occurrence < 2 observations) were excluded to avoid the disconnection of samples. Ordinations were performed using the 'metaMDS' function in package 'vegan' (Oksanen *et al.* 2013) in R.

Results

Camera-based survey

In total, 26 aquatic taxa were detected from the 18 cameras deployed, including 24 fish species, a crustacean (mud crab, *Scylla serrata* Forskal, 1775) and a reptile (estuarine crocodile, *Crocodylus porosus* Schneider, 1801). Yellowtail trumpeter, *Amniataba caudavittata* (Richardson, 1845), was recorded on the most cameras (44.44%). The mean number of taxa (identified to genus or species level) detected by each camera bait treatment was 3.8 ($n = 6$, 95% CI = -0.4, 7.7) for thawed Australian sardines, 3.7 ($n = 6$, 95% CI = 1.4, 6.3) for canned sardines, and 3.5 ($n = 6$, 95% CI = 0.7, 6.3) for unbaited treatments. In total, 18 taxa were recorded on cameras baited with thawed Australian sardines, 16 taxa were recorded on cameras baited with sardines and 13 taxa were recorded on unbaited cameras (Table 1). A one-way ANOVA reported no significant effect of camera bait treatment on number of taxa detected ($F_{2,15} = 0.018$, $P = 0.982$). The 12 baited cameras recorded a total of 13 taxa (11 fish species) that were not detected by the unbaited cameras, while the unbaited cameras recorded two fish species not recorded by the baited cameras (Table 1). Across the six replicates of each camera bait treatment, more than 80% of species richness was detected in the first 30 min of deployment (Fig. 2).

Four taxa were observed feeding on the canned sardine-baited cameras, while eight taxa were recorded feeding on the thawed Australian sardine-baited cameras (Table 1). These taxa were representatives from a diverse range of feeding guilds, including piscivores, opportunists, detritivores, omnivores, and zoobenthivores, according to available literature (Table 1). Canned sardine and thawed Australian sardine treatments detected the equal greatest number of opportunist and zooplanktivorous taxa, unbaited and canned sardine treatments recorded the equal highest number of zoobenthivores, while unbaited and thawed Australian sardine treatments recorded the equal highest number of piscivores. Thawed Australian sardine treatments recorded the greatest number of detritivores and

piscivores/zoobenthivores, while canned sardine treatments recorded the greatest number of herbivores/detritivores and omnivores. In cases where individual fish could be identified only to genus, the predominant feeding guild for this genus was assigned. A single, large adult estuarine crocodile was estimated to be 3.8 m long, based on scaling from head length relationships (see Fukuda *et al.* 2013) and a bait bag containing pilchards based on still video frames (Fig. 1d).

Multivariate analysis of species and functional group composition provides preliminary evidence that there are no substantial differences between the species captured in video samples under different bait treatments. Composition of video samples varied widely but showed no consistency in terms of the different bait treatments (Appendix 1). If there were important and consistent differences between treatments, then the points for each treatment would show separation in the 2D space.

In total, 18 h of video footage was collected and processed, with 22 instances where the single observer viewing the footage could not determine the classification of a species. For some instances, it was difficult to achieve a species-level identification from video alone, particularly for Ariidae species and juveniles of *Acanthopagrus*, *Gerres* and *Sillago*.

Postprocessing of video

Considerable visual changes in the clips were achieved using the algorithms (Fig. 3). Average confidence in identification by coauthors decreased with every shift in taxonomic level from order ($\bar{x} = 72\%$) to species ($\bar{x} = 17\%$) (Fig. 4a).

Treatment 2 (UDCP + CLAHE) resulted in the highest average identification confidence at the family ($F_{3,84} = 1.263$, $P = 0.046$), genus ($F_{3,84} = 1.263$, $P = 0.008$) and species ($F_{3,84} = 1.263$, $P = 0.016$) levels; however, there was no significant difference between treatments at the order level ($F_{3,84} = 1.263$, $P = 0.292$) (Fig. 4b). *Post hoc* analysis showed significant differences in mean identification confidence between Treatment 2 and Treatment 1 at the family (*t*-test, $t_{42} = -2.79$, $P = 0.008$) and genus level (*t*-test, $t_{42} = -3.04$, $P = 0.004$), and Treatment 2 and Raw at the species level (*t*-test, $t_{42} = -2.86$, $P = 0.006$). All other pairwise comparisons made at each taxonomic level were not significant. Overall, average identification confidence for clips processed with Treatment 2 (UDCP + CLAHE) was 25.2% greater than the average identification confidence for the raw, unprocessed clips.

Taxonomic resolution achieved with confidence values of $\geq 90\%$ varied by postprocessing treatment and taxonomic level (Fig. 5). Across all taxonomic levels, Treatment 2 (UDCP + CLAHE) produced the highest proportion of identifications made with $\geq 90\%$ confidence. Researchers viewing clips that had been postprocessed with Treatment 2 were able to make identifications to species level in four of the 22 clips with $\geq 90\%$ confidence, compared to just one for the raw footage and Treatment 1, and zero for Treatment 3. In total, targets in five of the 22 processed clips could be identified to species level by a panel of four researchers with the aid of a combination of algorithms. Of these five identifications to species level, two were of the same fish identified by different researchers using different treatments; therefore there were four unique identifications achieved to species level through postprocessing. These additional identifications to species level represent an overall

Table 1. Number of cameras on which each species was detected, across 18 baited and unbaited cameras
MaxN in shown in parentheses. Species are classified according to feeding guild (after Elliott *et al.* 2007) and bait feeding behaviour

Species	No. of deployments present (MaxN)			Bait feeding		References
	Unbaited	Canned sardines	Thawed sardines	Canned sardines	Thawed sardines	
Detritivore						
<i>Liza</i>	–	–	1 (1)	–	–	Pusey <i>et al.</i> (2004)
Herbivore/Detritivore						
<i>Scatophagus argus</i>	–	1 (2)	–	–	–	Gandhi (2002)
<i>Selenotoca multifasciata</i>	2 (2)	1 (2)	1 (4)	–	Yes	Thomson (1959)
Omnivore						
<i>Amniataba caudavittata</i>	2 (2)	1 (5)	2 (7)	Yes	Yes	Wise <i>et al.</i> (1994)
<i>Arrhamphus sclerolepis</i>	–	1 (1)	–	–	–	Waltham and Connolly (2006)
Opportunist						
<i>Scylla serrata</i> (Malacostraca)	–	2 (1)	2 (1)	Yes	Yes	Hill (1976)
Piscivore/Zoobenthivore						
Ariidae	–	1 (1)	1 (1)	–	Yes	Pusey <i>et al.</i> (2004)
<i>Crocodylus porosus</i> (Reptilia)	–	–	1 (1)	–	Yes	Taylor (1979)
<i>Lutjanus argentimaculatus</i>	1 (1)	–	2 (1)	–	Yes	Salini <i>et al.</i> (1990), Sheaves and Molony (2000)
<i>Terapon jarbua</i>	–	1 (1)	1 (3)	–	Yes	Whitfield and Blaber (1978), Baker and Sheaves (2005)
Piscivore						
<i>Scomberoides commersonianus</i>	2 (1)	–	1 (1)	–	–	Abrantes and Sheaves (2009)
<i>Sphyræna</i>	–	1 (1)	–	–	–	Blaber (1982)
<i>Caranx sexfasciatus</i>	1 (1)	–	–	–	–	Blaber and Cyrus (1983)
<i>Caranx</i>	–	–	2 (1)	–	–	Blaber and Cyrus (1983)
Zoobenthivore						
<i>Acanthopagrus</i>	1 (1)	1 (2)	2 (3)	Yes	Yes	Sheaves and Molony (2000)
<i>Drepane punctata</i>	–	1 (1)	–	–	–	Longhurst and Pauly (1987)
<i>Gerres filamentosus</i>	2 (13)	2 (2)	1 (5)	–	–	Wilson and Sheaves (2001)
<i>Gerres oyena</i>	3 (9)	2 (6)	1 (2)	–	–	Wilson and Sheaves (2001)
<i>Gerres</i>	1 (22)	–	–	–	–	Wilson and Sheaves (2001)
<i>Himantura uarnak</i>	1 (1)	–	2 (1)	–	–	Salini <i>et al.</i> (1990)
<i>Lutjanus fulviflamma</i>	–	1 (1)	–	–	–	Kamukuru and Mgaya (2004)
<i>Mesopristes argenteus</i>	1 (1)	1 (2)	–	–	–	Davis <i>et al.</i> (2011)
<i>Sillago</i>	1 (1)	1 (1)	–	–	–	Gunn and Milward (1985)
Zooplanktivore						
<i>Ambassis interruptus</i>	–	1 (70)	1 (25)	–	–	Pusey <i>et al.</i> (2004), Baker and Sheaves (2005)
<i>Pseudomugil</i>	–	1 (1)	–	–	–	Pusey <i>et al.</i> (2004)
<i>Monodactylus argenteus</i>	1 (1)	–	1 (1)	–	–	Nanjo <i>et al.</i> (2008), Sheaves and Molony (2000)
Total taxa	13	17	16			

improvement in species identification of 6.5% in the context of the current study.

Discussion

Postprocessing

The major contribution of the current study has been demonstrating that postprocessing by applying UDCP algorithms can aid in the identification of mobile aquatic taxa in low-visibility conditions. Additionally, the study tested the influence of bait and set time in rapidly surveying a tropical estuarine fish assemblage. Although scrutiny of bait effects is becoming

common in other aquatic systems (e.g. Harvey *et al.* 2007; Bassett and Montgomery 2011; Ebner and Morgan 2013), few studies have explored these issues in tropical estuaries. Further studies of fish assemblages in estuaries based on video detection have the potential to reveal novel aspects of fish assemblage composition (e.g. Sheaves and Johnston 2009) and to improve methods for visually assessing fauna in ecosystems compromised by suboptimal water clarity.

In the current study, increased confidence in visually identifying fishes was achieved through postprocessing of video footage. However, variable results were achieved in accordance with the application of different algorithms relative to each other

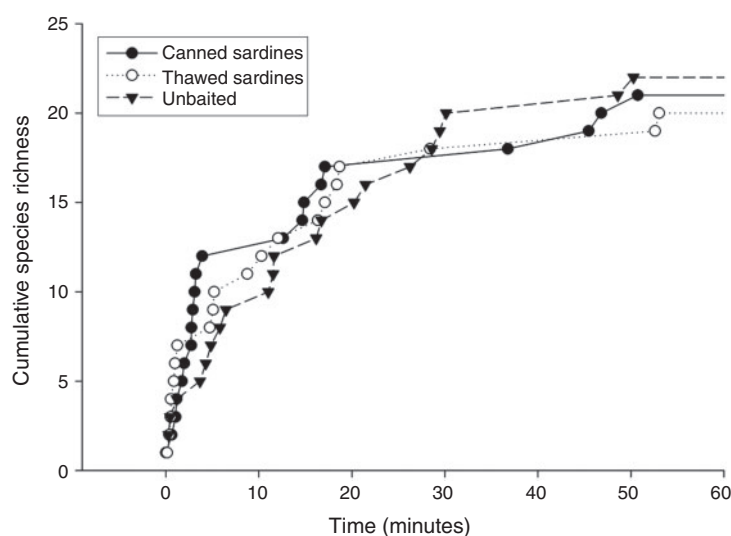


Fig. 2. Species accumulation curves for different camera bait treatments including BRUVS baited with canned sardines ($n = 6$), BRUVS baited with thawed Australian sardines ($n = 6$) and unbaited cameras ($n = 6$).

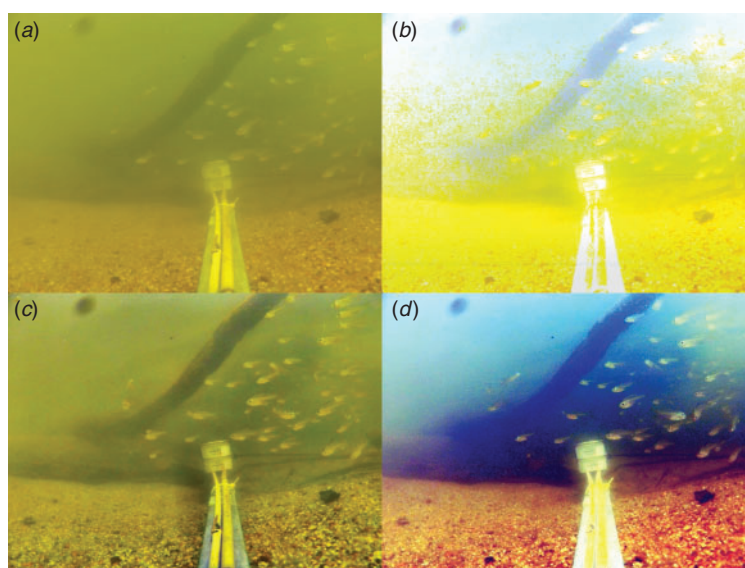


Fig. 3. Screenshots of the same clip from (a) original, unprocessed footage, (b) processed footage using Treatment 1 (UDCP and white balance), (c) Treatment 2 (UDCP + CLAHE), and (d) Treatment 3 (UDCP and histogram equalisation).

and raw footage. It would be interesting to conduct further tests of this kind within estuarine and other low-visibility aquatic ecosystems (e.g. nearshore coastal environments, rivers) based on capturing fishes near to cameras or under laboratory conditions where species identification is known. In many cases, choosing sampling occasions and locations based on an understanding of drivers and timing of best water clarity (e.g. seasonal, tidal cycles) may ultimately increase the number of reliable taxonomic identifications rather than relying heavily on post-processing algorithms. However, even when timing deployment

with likely best conditions, researchers visiting remote locations will usually have to make the best use of whatever data can be collected within the time available. In such instances, post-processing video may increase the amount of useable data when conditions are suboptimal. Alternatively, other studies have reported discarding footage once visibility is deemed to be insufficient (e.g. Broadhurst *et al.* 2006; Becker *et al.* 2012). At the time of publication, it is acknowledged that there have been technological advances in resolution of affordable underwater cameras (e.g. GoPro Hero 2 used here is now at model

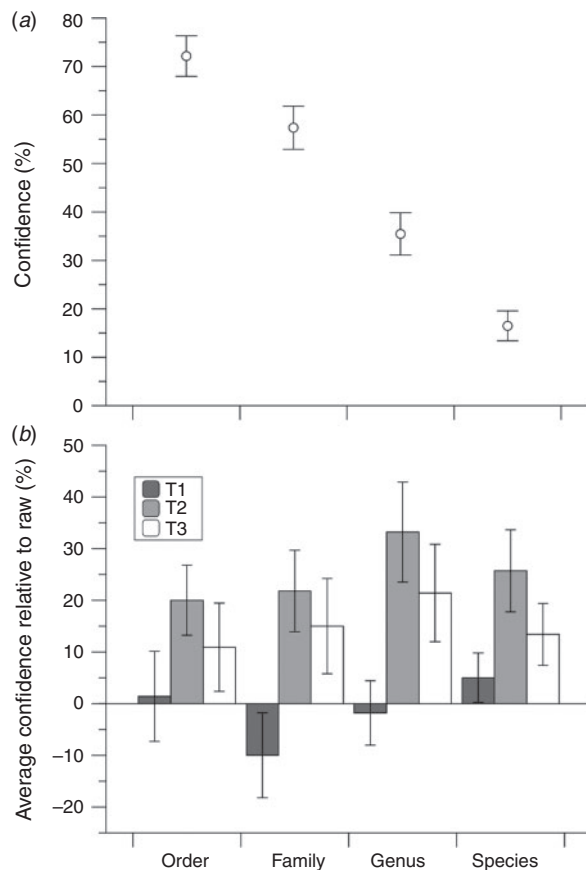


Fig. 4. (a) Mean confidence ($\pm$ s.e.) associated with taxonomic identifications at the order, family, genus and species level based on pooled data derived from 22 clips and four researchers; (b) mean confidence levels ($\pm$ s.e.) for postprocessing treatments 1–3 (T1 = UDCP and white balance; T2 = UDCP and CLAHE; T3 = UDCP and histogram equalisation) relative to the original, unprocessed clips.

Hero 8), and similarly there have been advances in video-processing algorithms. There are opportunities to further investigate synergistic benefits of improvements in these technologies for aiding identification of species in estuarine ecosystems.

Postprocessing may also be useful in answering particular types of ecological questions. Examples include studying shifts in animal behaviour and ecology under variable water clarity scenarios (e.g. when studying the arrival and passing of a river flood or an estuarine tidal cycle). We envisage this might be especially relevant when capture-based techniques (e.g. netting) are prohibitive or ineffective during periods of diminished visibility within cycles of interest (e.g. arrival of flood peaks, ebbing tides) or across spatially heterogeneous waterscapes (e.g. clear-water wetlands and more turbid systems). In which case, combining video and dual frequency identification sonar (DIDSON) (e.g. Becker *et al.* 2011) may represent a powerful synergy. Postprocessing of video may also have relevant implications for answering conservation questions where accurate surveys of a given community or assemblage are central.

Attractants

No clear difference was found in the average species richness detected by baited, as opposed to unbaited, cameras in the current comparison. While this finding has been reached in other studies, individual species appear to exist along a continuum from being attracted to or avoiding bait or bait-related activity (e.g. Dorman *et al.* 2012; Ebner and Morgan 2013). Conversely, Harvey *et al.* (2007) found that species richness estimates were increased with baited, as opposed to unbaited, cameras. In that study, large predators and scavengers were attracted to baited cameras in greater numbers than in the absence of bait. Herbivores were observed 4–7 m beyond the bait but remained essentially equally detected across baited and unbaited treatments. It is perilous to extrapolate from the findings of clear-water systems, such as the study of Harvey *et al.* (2007), to low-visibility systems and their fish assemblages in estuaries

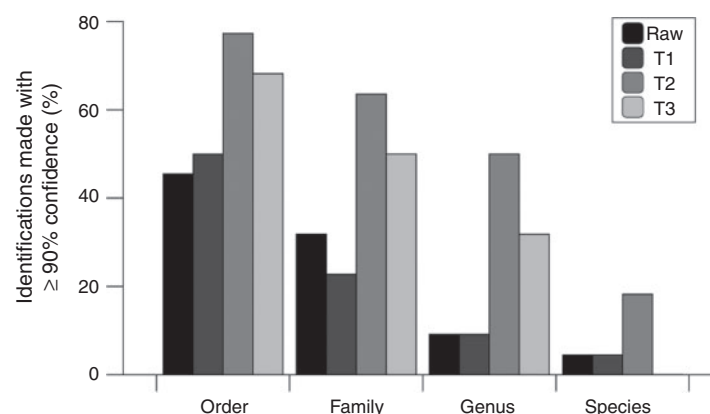


Fig. 5. Overall proportion of clips where identifications were resolved with $\geq 90\%$ confidence (self-assessed) by four researchers at the order, family, genus and species level. Data for the original, unprocessed footage (Raw) as well as three postprocessing treatments (T1 = UDCP and white balance; T2 = UDCP and CLAHE; T3 = UDCP and histogram equalisation) are presented.

with relatively poor visibility. The potential trade-off between attracting certain species to enhance their detection and identification is set against the possibility of repelling or not detecting shy species (Harvey *et al.* 2007). Furthermore, the fauna comprises different species and the assemblage is no doubt adapted to living in intermittent or consistently low-visibility conditions (in a diurnal sense) in many estuaries in northern Australia.

With regard to bait type, the current study indicated that there may be no clear benefit to using defrosted sardines over canned sardines or indeed over unbaited cameras for detecting species (richness). However, further investigation is required. This may have practical benefits for researchers working in remote tropical locations where storage of fresh or frozen baits is subject to significant logistical constraints. Interestingly, the number of instances where identification of a species was not possible for cameras baited with thawed Australian sardines (4) was $\leq 50\%$ that of unbaited (8) and canned sardines (10) deployments. This may indicate that this bait type brings species closer to the camera to enable a higher proportion of identifications. It is also important to note that, although there was little difference in average species richness detected by each of the bait treatments, the 12 baited cameras did detect 13 species not detected by the unbaited cameras. This result warrants further investigation through greater replication as it may have important implications for detecting a broader range of species from different feeding guilds. Combining baited and unbaited camera treatments may represent the optimal strategy for detecting species from a range of guilds within estuarine fauna assemblages. This 'hybrid' approach has also been suggested by previous studies in other aquatic settings (Watson *et al.* 2005; Ebner and Morgan 2013; Ebner *et al.* 2014).

Additionally, that a large estuarine crocodile attacked and removed the bait from one of the baited cameras in the current study without damaging the camera warrants brief discussion (see Fig. 1d). Studies from marine settings have previously reported issues with sharks removing bait bags (e.g. Langlois *et al.* 2006). Similarly, we have observed estuarine crocodiles attacking camera line floats and floats associated with nets, cameras and acoustic receivers in rivers and estuaries (Morgan, Ebner, Donaldson and Baker, pers. obs.) and freshwater crocodiles, *Crocodylus johnsoni* (Kreffelt, 1873), attacking camera baits in freshwaters (Ebner and Morgan, unpubl. data). This reinforces the need for a structural design that allows the camera to detach from the bait and emphasises the opportunity to investigate and ultimately minimise crocodile interaction with survey equipment. In this regard, it would be informative to test crocodile preference for types of floats or their alternatives, including short duration of camera set time (e.g. 15 min). Future designs may also incorporate biodegradable bait bags (e.g. hessian) to reduce the impact on large predators should they consume the bags.

Deployment time

With regard to the soak time of cameras, the current study found that species accumulation was similar among all three bait treatments, regardless of presence or absence of bait or bait type. For studies examining general assemblage composition, there is a case for using 30-min camera soak times based on the species accumulation curves shown in Fig. 2, since species addition is

much diminished from that point onward. This is comparable to a subset of studies from freshwater (e.g. Ebner and Morgan 2013) and marine (e.g. Harasti *et al.* 2015) systems. However, the generality of 30-min soak times would best be assessed with further data from a range of catchments, seasons, tidal cycles and magnitudes, and time of day. If rare species, including some sharks and rays, are the focus of a project it may be especially informative to compare long-deployment baited cameras with unbaited cameras, paying special attention to time of day and tidal cycle to quantify false negatives.

In summary, the effectiveness of underwater video in low-visibility conditions is enhanced by using a combination of baited and unbaited cameras to increase the probability of observing subsets of mobile estuarine fish assemblages, and postprocessing by applying UDCP algorithms can aid identification of mobile aquatic species.

Conflicts of interest

The authors declare no conflicts of interest.

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Appendix 1. Two-dimensional MDS ordination of (a) species (stress = 0.04) and (b) trophic functional group (stress = 0.09) composition of video samples with different bait treatments ($n = 15$)

Vectors show the direction and extent of correlation with the ordination space for (a) species and (b) trophic functional groups

