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Using remote underwater video to estimate freshwater fish species richness

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Species richness records from replicated deployments of baited remote underwater video stations (BRUVS) and unbaited remote underwater video stations (UBRUVS) in shallow (<1 m) and deep (>1 m) water were compared with those obtained from using fyke nets, gillnets and beach seines. Maximum species richness (14 species) was achieved through a combination of conventional netting and camera-based techniques. *Chanos chanos* was the only species not recorded on camera, whereas *Lutjanus argentimaculatus*, *Selenotoca multifasciata* and *Gerres filamentosus* were recorded on camera in all three waterholes but were not detected by netting. BRUVSs and UBRUVSs provided versatile techniques that were effective at a range of depths and microhabitats. It is concluded that cameras warrant application in aquatic areas of high conservation value with high visibility. Non-extractive video methods are particularly desirable where threatened species are a focus of monitoring or might be encountered as by-catch in net meshes.

Key words: baited video; diversity; feeding; fish community; rapid assessment; survey methodology.

INTRODUCTION

Especially in high visibility marine and estuarine systems, remote underwater video has been used to monitor fish population size, structure and abundance and the composition of fish assemblages and to observe fish behaviour without many of the problems associated with netting techniques (Holbrook & Schmitt, 2002; Cappo *et al*., 2007; Meynecke *et al*., 2008; Becker *et al*., 2010; Murphy & Jenkins, 2010). This has included application of baited remote underwater video stations (BRUVS) and unbaited remote underwater video stations (UBRUVS) (Priede & Merrett, 1996; Willis & Babcock, 2000; Cappo *et al*., 2004, 2007; Watson *et al*., 2005; Harvey *et al*., 2007; Heagney *et al*., 2007). BRUVSs were developed in part to overcome limitations associated with the maximum depth at which scuba diving could be used to perform visual census techniques (Cappo *et al*., 2007). While filming has been used to study fishes in freshwater ecosystems (Hinch & Collins, 1991; Irvine *et al*.,

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1991; Butler & Rowland, 2009), applications of UBRUVSs for monitoring fish assemblage composition in freshwater systems are rare (Ebner *et al*., 2009; Ellender *et al*., 2012) and applications of BRUVSs have not been tested, although it has been shown that the method works well for detecting crayfish (Fulton *et al*., 2012).

Surveys of inland fishes in remote areas of central and northern Australia have required the application of a range of traditional sampling gears (Puckridge *et al*., 2000; Unmack, 2001; Morgan *et al*., 2004; Morgan & Gill, 2004; Morgan, 2010; Kerezsy *et al*., 2011). This in part arises from the dynamic nature of the rivers and their waterhole dimensions, which can change seasonally or episodically, from one extreme of high-velocity flows in channels and overflow onto expansive floodplains through to the other extreme of isolated large or small waterholes, remnant small or large tributaries or even puddles during drying (Puckridge *et al*., 2000; Morgan *et al*., 2004; Beesley & Prince, 2010; Kerezsy *et al*., 2011). In recently flooded and expansive floodplain habitats, fish density can be low, whereas contraction of water levels back into waterhole refugia can lead to such high concentrations of fishes as to be prohibitive for replicated applications of sampling gear (Arthington *et al*., 2005).

Capture-based techniques, most commonly different types of netting, are often used to survey fishes in these systems, including beach seines, fyke nets, gillnets, dip-nets, net tows or trawls and angling with hook and line (Puckridge *et al*., 2000; Unmack, 2001; Morgan & Gill, 2004; Morgan *et al*., 2004; Arthington *et al*., 2005; Morgan, 2010; Kerezsy *et al*., 2011). Other capture techniques that have been used include application of rotenone or modified nets (*e.g*. drop-net) (Beesley & Gilmour, 2008). Electrofishing has been widely used to survey fishes in north-eastern and temperate Australia (Gehrke & Harris, 2000; Kennard *et al*., 2006; Stewart-Koster *et al*., 2007; Ebner *et al*., 2008). Direct observation from the surface or by snorkelling has also sometimes been used to survey freshwater fishes in Australian systems (Bishop *et al*., 1995; Hattori & Warburton, 2003; Morgan & Gill, 2004; Hardie *et al*., 2006; Ebner *et al*., 2009, 2011; Ebner & Thuesen, 2010; Thuesen *et al*., 2011), although turbidity and the threat of attack by crocodiles and bull sharks *Carcharhinus leucas* (Müller & Henle 1839) make underwater surveys undesirable in much of the tropics. Permanently turbid or sporadically turbid waterholes are characteristic of most catchments within central and northern Australia (McGregor *et al*., 2006), thwarting the widespread application of visual-based survey techniques. Under conditions of high visibility, however, visual techniques have the potential to be highly informative. Additionally, netting and electrofishing pose threats to a variety of species of special conservation importance. Platypus *Ornithorhynchus anatinus* and turtles can be drowned in mesh or fyke nets and harmed by electrofishing (Grant *et al*., 2004; Ebner *et al*., 2007). Crocodiles and large elasmobranchs such as sawfishes (*Pristis* spp.) are especially vulnerable to entanglement and drowning in mesh nets. Therefore, the choice of sampling method in their habitats must be a careful one.

It is proposed that BRUVSs and UBRUVSs will provide a versatile, non-extractive tool for surveying fishes in a wide range of microhabitats within large waterholes where turbidity is low. Using a relatively pristine river system in the undeveloped tropics of north-western Australia as a case study, the aim of this study was to (1) test different video deployment techniques, (2) compare species richness estimates based on remote underwater video (with and without bait) and conventional netting techniques and (3) explore strategies for combining different survey techniques in surveys of tropical fish communities. Additionally, the deployment of cameras in water depths of < or > 1 m was tested to determine if boat-based deployments were likely to be necessary in future surveys of fishes in the Fortescue River.

MATERIALS AND METHODS

STUDY AREA

Fishes were surveyed in three large waterholes (Table I) in the lower western Fortescue River catchment in the Pilbara region of Western Australia (Fig. 1) on 15–18 June 2010 during the dry season. The details of this catchment and the local climate are comprehensively described by Beesley & Prince (2010). Air temperature ranges from a minimum of *c*. 10° C in winter to >40° C in summer. The Fortescue River catchment is coastal but is situated in the Australian semi-arid zone and receives wet-season rainfall most typically from December to March; however, annual rainfall is highly variable. The Fortescue River often exists as a chain of waterholes. Large permanent waterholes contract to dimensions in the order of hundreds of metres to several kilometres in length, tens of metres in width and *c*. 10 m maximum depth.

CAMERA DEPLOYMENT AND RETRIEVAL

All survey methods were applied exclusively during daylight between 0800 and 1700 hours to avoid crepuscular or nocturnal effects of fish behaviour on species-specific detection (except for fyke nets). At each of the three waterholes, three UBRUVSs were set in wadeable depths (shallow water; <1 m depth) and three UBRUVSs were set in deep water (>1 m and up to 4·5 m) for a minimum of 1 h (and essentially all six cameras filmed simultaneously). This distinction between shallow and deep water has no ecological basis, and was used because most conventional netting gears that were to be used in this comparison of methods were applied to a depth of *c*. 1 m. UBRUVSs were placed essentially randomly to encounter a range of habitat types, and cameras were placed at least 50 m apart. The microhabitats that were commonly encountered included submerged macrophytes, bedrock and large boulders (sometimes covered in a green sponge), cobble–pebble and open sand; combinations of these microhabitats were often visible on individual cameras. A mud-silt benthos was occasionally encountered in the deep part of the most downstream waterhole surveyed. In especially dense macrophyte beds, cameras were lowered onto boulders or small clearings to prevent a totally inhibited field of camera view. A boat-based deployment of a UBRUVS is shown in Fig. 2. Occasionally, shallow-water cameras were positioned by wading from shore. UBRUVSs comprised a

TABLE I. Physical attributes of the three waterholes surveyed for fishes (see Fig. 1)

Variable	A	B	C
Distance to sea (km)	47·1	48·8	52·8
Altitude (m a.s.l.)	50	51	52
Surface area (m^2)	*c*. 68 000	*c*. 31 000	*c*. 30 000
Maximum length (m)	1140	780	860
Maximum width (m)	80	50	40
Maximum depth (m)	4·5	3·5	2·8
Vertical visibility (m)	3	3·5+	2·8+
Maximum horizontal visibility (m)	>5	>10	>10
Macrophyte cover (%)	20–30	20–30	20–30
Substratum	Cobble, sand, silt	Cobble, boulder, silt	Cobble, boulder, silt

FIG. 1. The location of the Fortescue River in Western Australia and the three waterholes (A–C) surveyed for fishes in this study and their approximate size at the time of sampling.

Sony HandiCam (HDRXR550; www.sony.com) set to a fixed focal length of infinity to avoid autofocus of objects or bubbles immediately on or near the lens (a wide-angle adapter was not used). Each camera was placed upright in a housing strapped to a half-hollowed concrete block (200 mm × 200 mm × 200 mm) (Fig. 2). The housing comprised two PVC pipes chemically fused in the middle and having threaded caps at either end. The rear end of the housing was a permanently closed screw-on cap fitted with an O-ring. The front part of the housing differed from the rear in only having a perspex window for the camera to view through. The housing was fitted with foam to enable a tight fit for the camera, but with adequate spaces to avoid camera overheating. A rope and buoy was used to lower the UBRUVS to the substratum and for shallow-water deployments cameras were faced away from direct sunlight to minimize glare.

UBRUVSs were retrieved by grabbing the float line and hoisting the camera onboard or occasionally by wading from shore if less disturbance would be caused by the latter approach (*e.g.* where the propeller would probably have churned up macrophyte beds). Following retrieval, each UBRUVS was converted into a BRUVS. Bait was attached and fixed at 1 m distance from the block base of the camera using cable ties and two parallel canes. A coarse mesh bait bag containing whole pilchards *Sardinops sagax* (Jenyns 1842), flathead grey mullet *Mugil cephalus* L. 1758 fillets and prawns *Penaeus latisulcatus* was overlaid on a fine-mesh bait bag containing similar bait (Fig. 2). The coarse mesh provided access to the bait for fishes, whereas the fine-mesh bait was intended to maintain an attractant to the camera field of view should the bait in the coarse-mesh bag become depleted. Bait was not pulped. BRUVSs were deployed at essentially the same position where each UBRUVS had been positioned in the waterhole. The operating time of each technique was recorded. Footage was downloaded each night to external hard-drives in the field.

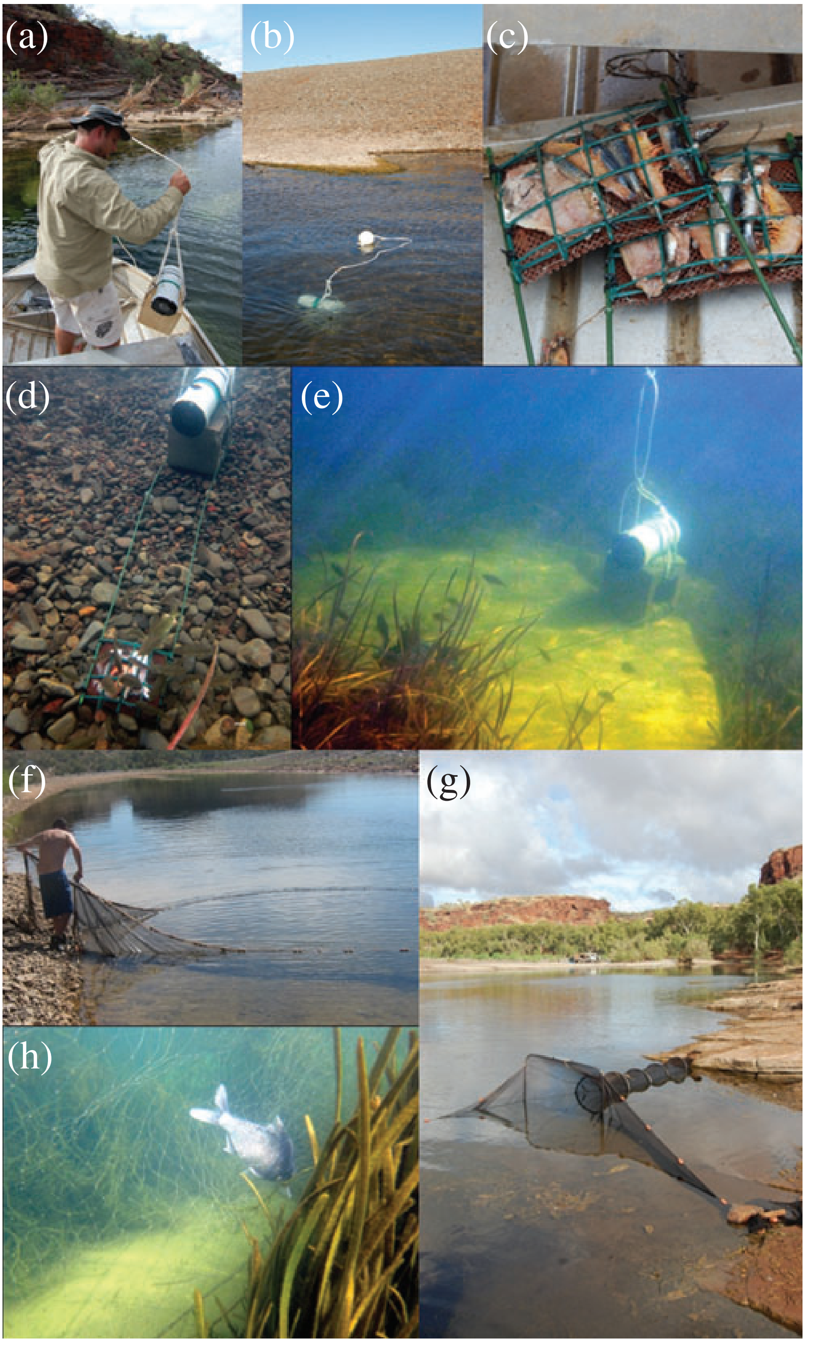

FIG. 2. Survey techniques including (a) deploying an unbaited remote underwater video station (UBRUVS) into deep water, (b) a UBRUVS deployed in shallow water, (c) the bait used in baited remote underwater video station (BRUVS) with whole pilchards *Sardinops sagax*, flathead grey mullet *Mugil cephalus* fillets and prawns *Penaeus latisulcatus* held in coarse mesh above (green) and in fine mesh below (brown) and supported by cane extension poles, (d) a BRUVS in shallow water, (e) a BRUVS in deep water, (f) beach seining, (g) fyke net and (h) a gillnet having snared a *Nematolosa erebi*.

CONVENTIONAL NETTING

Beach seining (26 m long with a 10 m pocket of 3 mm woven knotless mesh and two 8 m wings of 6 mm mesh and fished to a depth of 1·5 m) (Fig. 2) was conducted in shallow water at three locations within each waterhole, generally shadowing the earlier placements of cameras in shallow water, but allowing opportunity (*c*. 1 h) for fish to disperse (there was no obvious sign that fishes remained congregated in these areas following removal of baited cameras). Similar spacing was used for three fyke nets (11·2 m in width, with two 5 m wings and a 1·2 m wide mouth, depth of 0·8 m, all comprising 2 mm woven mesh) (Fig. 2), which were set in shallow water between 1700 and 1830 hours and retrieved between 0800 and 1000 hours. This was the only method used during the night in this study and mimicked typical applications of fyke nets to survey Australian freshwater fishes (Puckridge *et al*., 2000; Kerezsy *et al*., 2011). A multi-panel gillnet (50 m in length comprised equally of 50 and 100 mm monofilament mesh) (Fig. 2) was set (floating) perpendicular to shore for 15 min in each of the three locations during daylight hours. Gillnets (Fig. 2) were always set after camera-based techniques were completed to avoid any bias associated with using a potentially destructive sampling technique first (the reverse effect of baits having drawn fishes into the netting areas was considered to be negligible). All fishes were identified using the method described by Allen *et al*. (2002), and those caught using nets were also counted and measured (total length, L_T). A sub-sample of hybrid terapontid specimens was also retained to confirm taxonomy. The operating time of each technique (including counting and measuring catches, and saving and processing video) was recorded.

VIDEO PROCESSING

No attempt to measure fish length was made in relation to camera-based techniques (although the bait bags provide a useful scale to achieve crude estimates, if necessary). Video was viewed in VLC Media Player (VideoLan Team; www.videolan.org/videolan/team) on a MacIntosh computer (www.apple.com) in real time for the first 10 min of each deployment. The following 50 min was then viewed in a mixture of real time or fast forward (×1·5 or ×2 real time speed) depending on a number of factors (*e.g*. how diverse the fish assemblage was on a particular camera and underwater visibility). Record was made of the first arrival of each species in view on each deployment, and of N_{max} [the maximum number of each species in view following Willis & Babcock (2000) and Cappo *et al*. (2004)] in the first 10, 20, 30, 40, 50 and 60 min of each deployment. Multiple viewing of film was required in certain cases (*e.g*. high species richness, high densities of fish, presence of small fish or difficult to identify fish; Fig. 3). N_{max} was estimated in increments of five in cases where an exact count was unachievable (*i.e*. >15, >20 ... >30) and as >100 and >200 for larger values. These values were eventually registered as the minimum values (*e.g*. >15 = 16, >200 = 201). A single operator observed the film except where identifications were difficult, in which case consensus was sought. In cases where the identification of the species remained uncertain, the fish was excluded from the analysis.

DATA ANALYSES

The frequency of occurrence of species across replicate deployments of survey techniques (*i.e*. presence or absence of each species across all waterholes pooled, *e.g*. beach seines $n = 9$) was used to (1) compare the effectiveness of each technique in contributing to measuring species richness and (2) determine species-specific detection probabilities underlying species richness estimates. Average catch or average N_{max} was calculated in relation to each sampling technique to determine if video-based techniques provide comparable count data (in terms of means and variance) to conventional netting catch data.

Comparison of video-based techniques was conducted to aid future applications of this relatively new approach in the context of freshwater systems. Specifically, species accumulation curves were plotted based on arrival time (first detection) of each species on a camera according to technique. Shallow-water UBRUVS is referred to as sUBRUVS, deep-water UBRUVS as dUBRUVS, shallow-water BRUVS as sBRUVS and deep-water

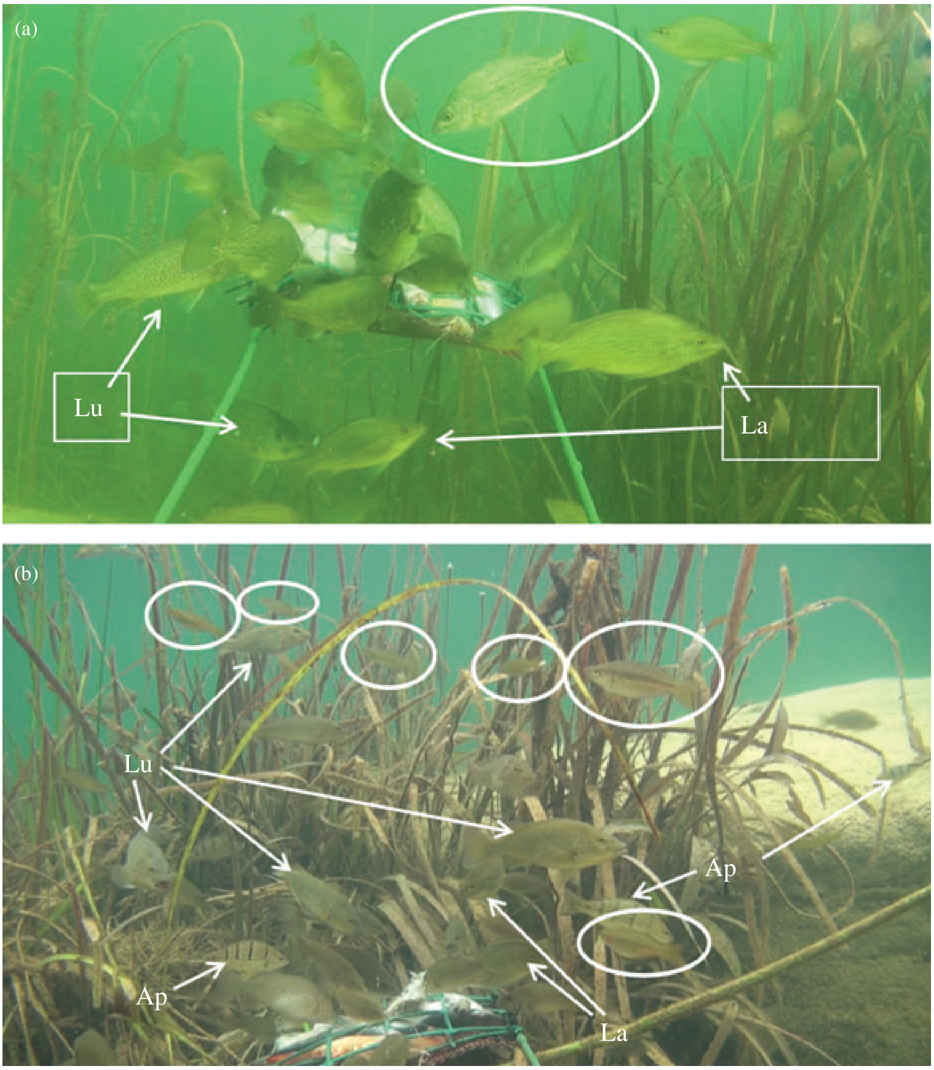

Fig. 3. Video frames of (a) striped hybrid terapontid (encircled) at a baited remote underwater video station (BRUVS) with examples of *Leiopotherapon unicolor* (Lu) and *Leiopotherapon aheneus* (La) and (b) vertical positioning of fish species in relation to field of view at a shallow-water BRUVS, with *Melanotaenia australis* (Ma) (encircled) mostly near to the water surface, and grunter species [Lu, La and *Amniataba percoides* (Ap)] more closely associated with the benthos and in close proximity to the bait.

BRUVS as dBRUVS. Species richness in each waterhole was then recalculated by simulating if the cameras had been deployed for 10, 20, 30, 40, 50 or 60 min to determine optimum camera deployment time. Average time until arrival of each species was calculated from the sub-set of data where species were detected on camera (*i.e.* a value was not included for a species if that species was not detected on a particular camera).

It would not be unexpected that shallow-water techniques (*e.g.* seining and sUBRUVS) are more likely to encounter shallow-water species than deep-water techniques (*e.g.* gillnetting and dUBRUVS), and that fyke-netting is more likely to encounter nocturnal species compared with other methods. Further, it would be expected that BRUVSs, by virtue of being associated with bait, would potentially attract a higher proportion of carnivores than UBRUVSs. To compare the species captured with each different sampling method and in the different habitat types, such as shallow or deep water, the presence or absence data for each individual replicate for each method was used to construct a similarity matrix using the Bray–Curtis similarity coefficient in the PRIMER package, once it was allocated to a method (*e.g.* seine, fyke, gill, dUBRUVS, sUBRUVS, dBRUVS or dUBRUVS) (Clarke & Gorley, 2001). Statistical differences were generated by subjecting the matrix to one-way ANOSIM (Clarke & Gorley, 2001). ANOSIM generates an *R*-statistic which is an estimate of the similarity of the replicates within these predetermined methods (*e.g.* fyke nets, seines, gillnets, sUBRUVS, dUBRUVS, sBRUVS and dBRUVS), with higher values (close to 1) suggesting that replicates within groups were more similar to each other than to any replicate within another group, while lower *R*-values (close to 0) suggest that there were similarities between and within groups (methods). For illustrative purposes, samples from each method within each waterhole were averaged and a classification plot was generated using the PRIMER package.

RESULTS

FISH ASSEMBLAGES

In total, 2377 fish observations were made (including video-based observations and net captures) comprising 14 species (Table II) from 12 families. Eight were primarily freshwater species and six were considered marine and estuarine species (Morgan & Gill, 2004), with the former recorded in all three waterholes (Table II). The number of marine and estuarine species diminished with increasing distance from the estuary (six, four and two species in waterholes A, B and C, respectively). Additionally, 69 observations of what appear to be hybrid terapontids were recorded including 64 of a striped form (Fig. 3) and five of a spotted form (Table II). No other aquatic vertebrates such as turtles were collected or observed, but 59 palaemonid shrimps (unidentified shrimps of <30 mm in length) were collected in nets.

SURVEY TECHNIQUE AND SPECIES RICHNESS ESTIMATES

Collectively, the camera-based techniques outperformed the conventional netting approaches, with the former yielding 13 species and the latter 11 species. Within each of the three waterholes, each of the netting techniques detected fewer species (five to seven species) than each of the camera-based techniques (10–11 species) (Table III). Three species [mangrove jack *Lutjanus argentimaculatus* (Forsskål 1775), spotbanded scat *Selenotoca multifasciata* (Richardson 1846) and whip-fin silver-biddy *Gerres filamentosus* Cuvier 1829] were recorded from video filming in all three waterholes and were not detected by conventional netting techniques (Figs 4 and 5). Only one species [milkfish *Chanos chanos* (Forsskål 1775)] was not recorded on camera, but was caught in gillnets (Fig. 4).

Time required to achieve three replicates of each technique at a waterhole was in the order of 1·5–2·5 h for all techniques except fyke nets that were set overnight and required 16–18 h (Table III). Time required for beach seining or gillnetting was accumulated linearly with replication, whereas other techniques (*i.e.* fyke nets and camera-based techniques) could be set and retrieved as batches. This is important to remember in comparisons of strategies involving a mixture of survey techniques as increasing replication of certain units (primarily passive gears, *e.g.* cameras

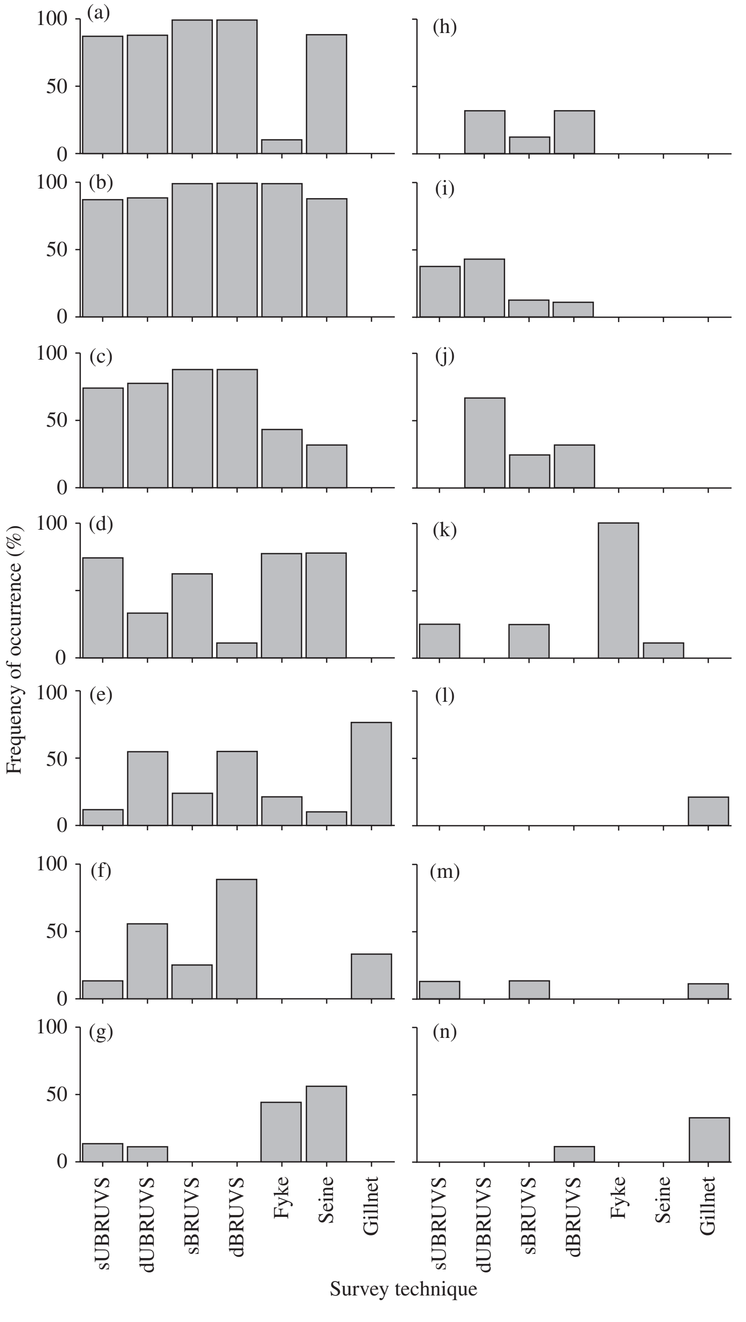

FIG. 4. Species detection (as per cent frequency of occurrence) per gear replicate (sUBRUVS, shallow water unbaited remote underwater video station; dUBRUVS, deep-water UBRUVS; sBRUVS, shallow-water baited remote underwater video station; dBRUVS, deep-water BRUVS), with gear replicates pooled across waterholes: (a) *Amniataba percoides*, (b) *Leiopotherapon aheneus*, (c) *Leiopotherapon unicolor*, (d) *Melanotaenia australis*, (e) *Nematalosa erebi*, (f) *Neoarius graeffei*, (g) *Glossogobius giuris*, (h) *Selenotoca multifasciata*, (i) *Gerres filamentosus*, (j) *Lutjanus argentimaculatus*, (k) *Neosilurus hyrtlii*, (l) *Chanos chanos*, (m) *Mugil cephalus* and (n) *Megalops cyprinoides*.

TABLE II. Total number of fish detections in the rapid survey of the three waterholes based on all techniques combined (see Fig. 1)

Species	A	B	C	Total
Freshwater species				
Amniataba percoides	78	58	61	197
Leiopotherapon aheneus	71	243	254	568
Leiopotherapon unicolor	32	68	72	172
Hybrid terapontid (5–7 stripes)	9	25	30	64
Hybrid terapontid (spots)	–	1	4	5
Melanotaenia australis	93	400	241	734
Neoarius graeffei	26	12	15	53
Nematalosa erebi	27	217	3	247
Glossogobius giuris	12	6	4	22
Neosilurus hyrtlii	7	117	89	213
Marine species				
Megalops cyprinoides	7	–	–	7
Chanos chanos	3	–	–	3
Mugil cephalus	16	5	–	21
Lutjanus argentimaculatus	12	2	4	18
Gerres filamentosus	–	6	7	13
Selenotoca multifasciata	23	17	–	40
Total	416	1177	784	2377

and fyke nets) requires minimal additional time and effort, whereas others (primarily active gears, *e.g*. beach seines; but also short soak times of gillnets) incur greater cost.

Duration in the order of 3 h was also required to check footage quality and make a copy of video recording each night in the field. Recording of the time of first arrival of each species and N_{max} corresponding to each camera deployment required a mean ± S.E. = 43 ± 2 min from UBRUVS and 45 ± 1 min from BRUVS.

TABLE III. Species richness estimates per survey technique and waterhole (see Fig. 1). Estimates of minimum time to achieve three replicates at a waterhole are also provided

Technique	A	B	C	Total	Field time (h)
Beach seine	7	4	5	5	1·5–2·5
Fyke net	6	6	6	7	16–18
Gillnet	5	1	2	5	1·5
Shallow UBRUVS	3	6	8	10	1·5
Shallow BRUVS	6	5	9	11	2
Deep UBRUVS	8	9	7	10	1·5
Deep BRUVS	8	9	8	10	2
All techniques combined	13	12	12	14	20

BRUVS, baited underwater video station; UBRUVS, unbaited remote underwater video station.

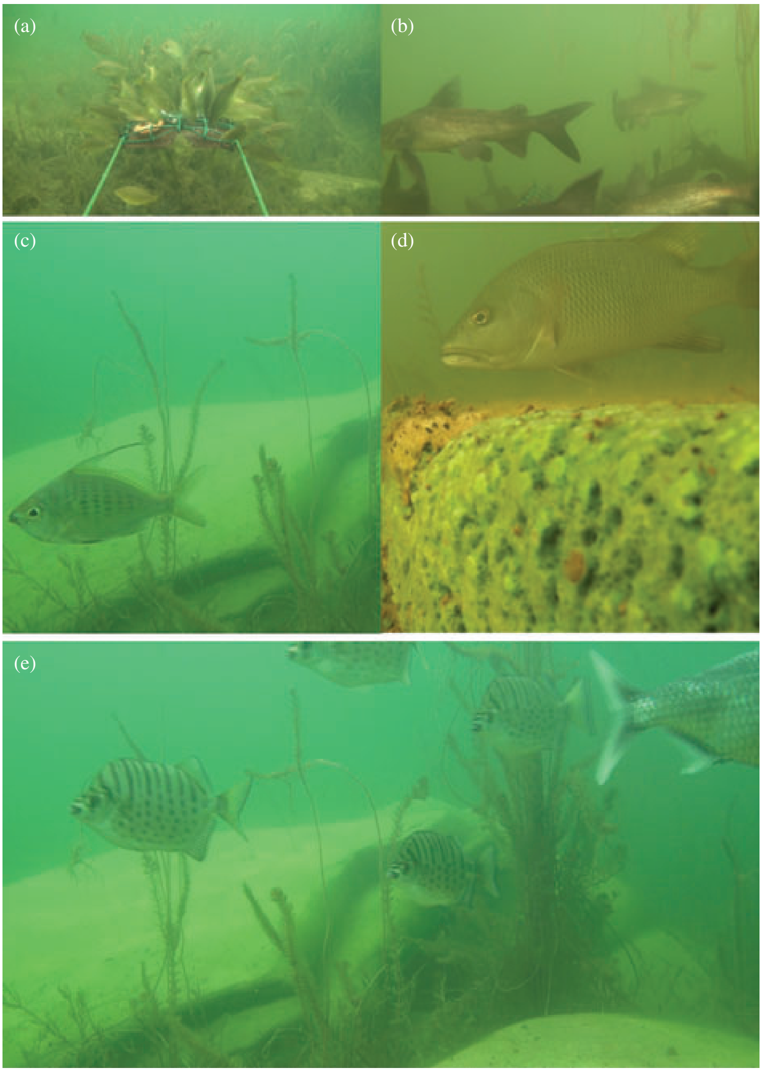

FIG. 5. Images of selected fishes recorded on underwater cameras: (a) large numbers of terapontids feeding at a baited remote underwater video station, (b) *Neoarius graeffei* at 3–3·5 m depth; and three species (c) *Gerres filamentosus*, (d) *Lutjanus argentimaculatus* and (e) *Selenotoca multifasciata* that were only detected by underwater camera techniques.

COMPARISON OF VIDEO-BASED TECHNIQUES

All video-based techniques recorded similar overall species richness at the among-waterhole scale (10–11 species), whereas patterns were unclear at the within-waterhole scale (Table IV). If camera techniques were combined, then (1) sBRUVSs and dBRUVSs marginally outperformed (2) sBRUVSs and sUBRUVSs and (3) dBRUVSs and dUBRUVSs, in terms of species richness estimation at the within-waterhole scale (Table IV). The combination of sUBRUVSs and sBRUVSs yielded relatively low estimates of species richness by waterhole, despite achieving a high among-waterhole species richness estimate. The majority of species captured on video were detected within 10 min of deployment, regardless of the camera technique used (Fig. 6).

SPECIES-SPECIFIC DETECTION

ANOSIM suggested that there were significant differences between all pair-wise comparisons amongst the species recorded by the different methods with the exception of sUBRUVSs compared with both sBRUV and seines, and dUBRUV compared with both sBRUV and dBRUVSs (Table V and Fig. 7). The relative probability of detecting each species was a function of survey technique (Fig. 4). The three Terapontidae species [barred grunter *Amniataba percoides* (Günther 1864), Fortescue grunter *Leiopotherapon aheneus* (Mees 1963) and spangled perch *Leiopotherapon unicolor* (Günther 1859)] were frequently recorded by camera-based techniques [Figs 4(a)–(c) and 5(a)]. Conventional netting also caught terapontids [Fig. 4(a)–(c)]. For example, fyke nets regularly captured *L. aheneus* and to a lesser extent *L. unicolor*, but were poor at detecting *A. percoides* [Fig. 4(a)–(c)]. Beach seining detected all of these grunter species, but inconsistently detected *L. unicolor* [Fig. 4(a)–(c)] and *L. unicolor* was not detected by any of the three beach seining replicates in waterhole A. Terapontids, however, were not collected in gillnets [Fig. 4(a)–(c)].

Western rainbowfish *Melanotaenia australis* (Castelnau 1875) was detected most successfully by shallow-water camera techniques, beach seine and fyke net [Fig. 4(d)], indicating that this species does not inhabit deep water [Fig. 3(b)]. Australian river gizzard shad *Nematalosa erebi* (Günther 1868) was best detected by deep-water BRUVS and to a lesser extent by deep-water UBRUVS in all the three waterholes [Figs 4(e) and 5(b)]. Blue salmon catfish *Neoarius graeffei* (Kner & Steindachner 1867) was most frequently detected by gillnet and to a lesser extent by deep-water camera techniques [Fig. 4(f)]. Gillnets detected *N. graffei* in two of the three waterholes [Fig. 4(f)]. Flathead goby *Glossogobius giuris* (Hamilton 1822) and Glencoe tandan *Neosilurus hyrtlii* Steindachner 1867 were best surveyed by conventional netting, either by beach seine or fyke net [Fig. 4(g), (k)]. Three species [*C. chanos*, *M. cephalus* and tarpon *Megalops cyprinoides* (Broussonet 1782)] were rarely encountered, but in these cases detection was by gillnet or sporadically by different camera-based techniques [Fig. 4(l)–(n)]. Additionally, three species were detected only by camera [*G. filamentosus*, *L. argentimaculatus* and *S. multifasciata*; Figs 3 and 5(c)–(e)]. Despite few observations regarding

TABLE IV. Species richness estimates from a combination of camera techniques. Estimates of minimum time to achieve three replicates at each waterhole are also provided (see Fig. 1)

Strategy	A	B	C	Total	Field time (h)
UBRUVSs only	9	9	9	12	2
BRUVSs only	9	10	11	12	2·5–3
Shallow BRUVSs and UBRUVSs	7	6	10	12	2·5
Deep BRUVSs and UBRUVSs	9	9	10	11	2·5
All camera techniques	11	10	12	13	4

BRUVS, baited underwater video station; UBRUVS, unbaited remote underwater video station.

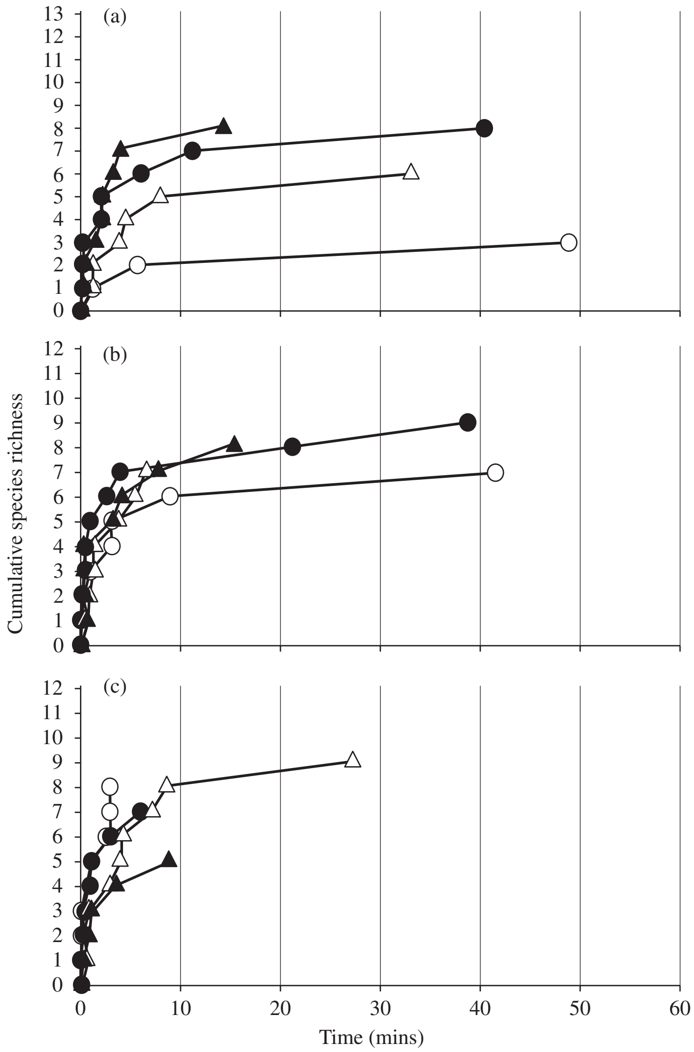

FIG. 6. Species accumulation curves with increasing viewing time in association with applying different camera-based techniques [sUBRUVSs, shallow-water unbaited remote underwater video stations (○); dUBRUVS, deep-water UBRUVS (●); sBRUVSs, shallow-water baited remote underwater video stations (△); dBRUVS, deep-water BRUVSs (▲)] in each of three waterholes: (a) A, (b) B and (c) C (see Fig. 1). The *y*-axis is scaled to the known species richness recorded by all survey techniques in each waterhole.

these species, indications were that *S. multifasciata* is more likely to be detected in deep rather than shallow water, and that *G. filamentosus* was mainly observed on unbaited cameras regardless of depth [Fig. 4(h), (i)]. *Lutjanus argentimaculatus* was more common in deep than shallow water [Fig. 4(j)]. This species was also as frequently detected on UBRUVS in deep water as it was on all of the other three types of camera techniques combined [Fig. 4(j)]. Both *G. filamentosus* and *L. argentimaculatus* appeared to be wary of the camera; specifically, *G. filamentosus*

TABLE V. *R*-values for ANOSIM pair-wise comparisons of the fish detections based on presence or absence data (species × abundance matrix) using the different methods in the waterholes of the Fortescue River, *i.e.* waterholes A, B and C. Significantly different relationships are represented by *, $P < 0.05$, **, $P < 0.01$ and ***, $P < 0.001$

	Shallow UBRUVSs	Deep UBRUVSs	Shallow BRUVSs	Deep BRUVSs	Fyke net	Seine
Deep UBRUVSs	0·156*	–	–	–	–	–
Shallow BRUVSs	−0·039	0·069	–	–	–	–
Deep BRUVSs	0·408***	0·054	0·255**	–	–	–
Fyke net	0·416***	0·753***	0·626***	0·919***	–	–
Seine	0·095	0·409***	0·213**	0·665***	0·480***	–
Gillnet	0·858***	0·765***	0·881***	0·831***	0·907**	0·900***

BRUVS, baited underwater video station; UBRUVS, unbaited remote underwater video station.

usually kept its distance from the camera and *L. argentimaculatus* sometimes briefly approached the camera, but spent little time in its view.

DISCUSSION

This study found that application of remote underwater video techniques enhances capabilities for surveying a freshwater fish community relative to exclusively using conventional netting techniques. Specifically, in terms of maximizing species richness estimates, applications of BRUVSs and UBRUVSs outperformed conventional netting techniques in the Fortescue River. Similarly, Ellender *et al*. (2012) found that unbaited cameras more effectively detected two threatened freshwater fishes in headland streams than electrofishing.

COMPARING SURVEY TECHNIQUES

This study demonstrates, as have numerous others (Watson *et al*., 2005), that no single sampling technique is adequate for detecting all fishes, and indeed most techniques were different in the species that they detected. Certain techniques were better all-rounders, maximizing species richness estimates (*e.g*. camera-based techniques), whereas others were specialised in effectively detecting a sub-set of species (*e.g*. fyke nets detected nocturnal *N. hyrtlii* far better than any other technique). The conventional netting techniques in combination provided a moderately sound strategy for surveying the fish assemblage in each of the three waterholes (see strategy 5 in Table VI) and this represents a relatively common suite of techniques used to survey fishes in waterholes in remote areas of central and northern Australia (Puckridge *et al*., 2000; Unmack, 2001; Morgan & Gill, 2004; Morgan *et al*., 2004; Morgan, 2010; Kerezsy *et al*., 2011). The major limitation of conventional netting, however, was the failure to detect three of the 14 species (*i.e. L. argentimaculatus*, *S. multifasciata* and *G. filamentosus*) encountered in this study (Table VI), whereas the remote video strategy detected these three species in all three waterholes (Fig. 4). As these species are often associated with estuaries or lower reaches of catchments,

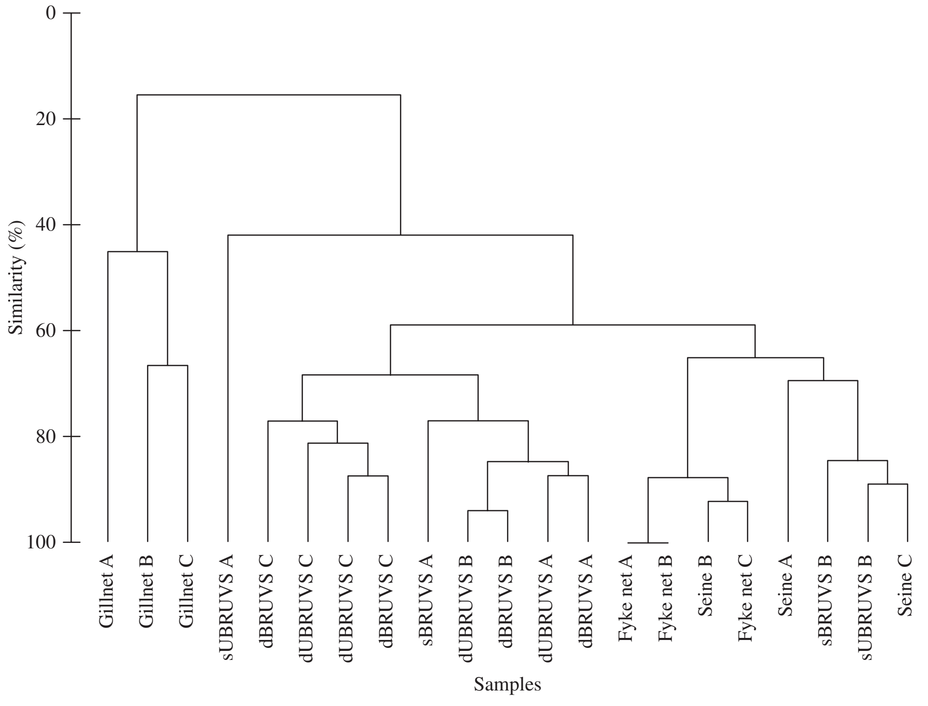

FIG. 7. Classification of the species recorded in each waterhole (denoted as A, B and C) using different sampling techniques (sUBRUVS, shallow-water unbaited remote underwater video station; dUBRUVS, deep-water UBRUVS; sUBRUVS, shallow-water baited remote underwater video station; dBRUVS, deep-water BRUVS). Resemblance: S17 Bray-Curtis similarity.

TABLE VI. Species richness estimates arising from combining certain survey techniques, and estimates of minimum time to achieve three replicates at a waterhole (see Fig. 1)

	Species richness by waterhole				
Strategy	A	B	C	Total	Field time (h)
(1) BRUVS only	9	10	11	12	2·5–3
(2) Seine and gillnet	9	7	6	11	3–4
(3) Camera-based techniques	11	10	12	13	4
(4) Seine, gillnet and BRUVS	12	12	11	14	5–6
(5) Conventional netting	11	7	8	11	16–18
(6) All techniques combined	13	12	12	14	20

BRUVS, baited underwater video station; UBRUVS, unbaited remote underwater video station.

it may be that there is much benefit in using underwater video at lowland sites to bolster fish species richness estimates in rivers of the Pilbara. Conventional netting detected one species (*C. chanos*) that remote video did not (Fig. 4), but this was based on few data (just two individuals collected in a single waterhole) and therefore has questionable relevance in a comparison of survey techniques.

One of the more useful attributes of underwater video in terms of assessing freshwater fish assemblages is its flexibility with regard to the range of habitat types in which it can function. Cameras can be placed in deep (including to abyssal depths; Priede & Merrett, 1996) and shallow water and in open water within or adjacent to relatively dense or complex structure (Becker *et al*., 2010). This is advantageous relative to many conventional netting techniques. For instance, beach seines are generally restricted to shallow water and smooth substrata and applications can be compromised by large woody structure and complex benthic topography; fyke nets are generally restricted to shallow water or require exclusion devices to prevent mortality of air-breathing species (*e.g*. amphibians, reptiles, birds and mammals) and this in turn excludes large-bodied fishes; gillnets fair poorly in structurally complex habitat and in fast-flowing environments and require monitoring to minimize mortality of target species and by-catch (Connolly, 1994; Sindilariu *et al*., 2002; Grant *et al*., 2004; Hardie *et al*., 2006; Ebner *et al*., 2007).

In the context of clear-water freshwater ecosystems, electrofishing and underwater visual census (UVC) probably represent the only methods with comparable flexibility relative to underwater cameras in terms of the range of habitats that can be surveyed for fishes, although electrofishing is not routinely applied to the extreme depths that cameras can access. Several studies have compared UVC and remote underwater video for enumerating fishes in marine systems and highlight the different views of community composition obtained by these methods (Willis & Babcock, 2000; Cappo *et al*., 2007), whereas comparisons of fish community assessments based on electrofishing, UVC and remote underwater video remain to be advanced in fresh waters.

An exception is the study of Ellender *et al*. (2012) that provided a comparison of the detection of two fishes by UBRUVSs and backpack electrofishing in shallow headwater streams in South Africa. The study showed that UBRUVSs were an effective technique for recording the relative abundance of two species relative to electrofishing, the latter being widely used in isolation to perform fish community assessment in shallow streams of several continents. This study demonstrates that UBRUVSs can be used to survey moderately species-rich fish communities in large waterholes. Comparisons of species' relative detection rates based on camera methods and boat electrofishing in deeper systems with more species is an obvious next step. In that context, baited cameras may prove useful for detecting highly mobile, low-density predators.

In this study, cameras were essentially used to target benthic fish communities, which may partly explain the lack of detection (*e.g. C. chanos*) or infrequent detection of certain open-water specialists (*e.g. M. cyprinoides* and *M. cephalus*). Remote underwater video has been used in marine systems to monitor fishes occupying midwater and pelagic habitats (Heagney *et al*., 2007). Similarly, application of floating cameras or cameras fixed at the water surface warrants investigation in freshwater ecosystems. Presumably, *M. australis* counts would have been greatly elevated in this study, if shallow-water cameras were oriented to view the water surface in addition to the benthos, as unlike most fishes, *M. australis* were usually observed in the top third of the field of view in this study [B. C. Ebner, pers. obs.; Fig. 3(b)].

VIDEO AND SPECIES RICHNESS ESTIMATION

The underwater cameras provided a useful means of identifying depth-related structuring of the fish assemblages. Specifically, a sub-set of species (*e.g*. terapontids and *G. filamentosus*) were frequently detected in both shallow and deep water, whereas others were more likely to be found in either shallow (*e.g. M. australis*) or deep water (*e.g. N. graeffei*). This indicates that the placement of survey gear with respect to depth will affect estimates of fish assemblage composition. One drawback of the cameras is their reliance on good visibility. Beesley & Gilmour (2008) obtained good estimates of species richness in the Fortescue River based on trials of a modified drop-net. Drop-nets do not rely on clear water, and a study conducted in waterholes of the Fortescue River found that this method provided a good description of the fish assemblage (Beesley & Gilmour, 2008). While drop-nets may be more widely applicable than video in Australian waterholes, they are labour-intensive in the field. For example, 3 h of effort is required for three replicate drops (Beesley & Gilmour, 2008). Camera techniques are far more efficient; for example, this study found that boat-based deployment and retrieval of six cameras can be achieved in <100 min (assuming 60 min filming time). If camera filming time is decreased to 20 min [appropriate for maximal species detection (Fig. 6)], then cameras will be even more efficient. Time saved could be used to increase within-waterhole replication or relaxing amidst a busy field schedule. Similar conclusions have been made from surveys of marine fishes (Stobart *et al*., 2007).

Surprisingly, BRUVSs did not attract fishes into view more rapidly than UBRUVSs in this study, indicating that bait may be unnecessary for fish surveys at least in the context of the Fortescue River. Baited and unbaited cameras did, however, yield subtly different types of information with regard to species richness. Unbaited cameras performed better than baited cameras in detecting the presence of some species in a waterhole (*G. filamentosus* and *G. giuris*) or the number of cameras that a species was detected on (*e.g. L. argentimaculatus*). The reverse was also the case for a sub-set of species (*e.g. M. cyprinoides* and *N. graeffei*), indicating that each method has particular biases making cross-study comparisons difficult and standardization of methods important. Harvey *et al*. (2007) found that herbivorous and omnivorous species were just as likely to be detected by UBRUVS as BRUVS, and that predators were more detectable by BRUVS than UBRUVS. Several investigations (Watson *et al*., 2005; Harvey *et al*., 2007; Stoner *et al*., 2008) have found that large predatory fishes are more likely to be detected by BRUVS than UBRUVS. Whilst similar results were found for a number of the predators encountered in this study (*N. graeffei*, *L. aheneus*, *L. unicolor* and *A. percoides*), the finding that the top predator *L. argentimaculatus* is best surveyed in deep water without bait is counter-intuitive. Potentially, this species is attracted to live prey far more than dead bait. *Lutjanus argentimaculatus* appears to be wary of cameras and of sampling gears in general, and its relative abundance has probably been underestimated by previous field surveys based on conventional netting.

This study has focussed on field-based survey methodology and its effects on species richness estimation. Underwater video-based methods have the potential to record additional information relating to natural behaviour, biology or ecology of fishes. No attempt was made to measure fish length in the field of view (Harvey *et al*., 2007) or make a detailed record of fish behaviour (Holbrook & Schmitt, 2002) in this study in the interests of brevity and in an effort to minimize video processing time in the laboratory (Ebner *et al*., 2009). Interesting fish behaviours were, however, noted. For example, *L. aheneus* often browsed on the surface of submerged macrophytes and both that species and *A. percoides* browsed from hard substrata, whereas

L. unicolor were not observed feeding (except on baited cameras). *Gerres filamentosus* fed by sifting through sediments, and *L. argentimaculatus* was at times solitary and at other times in schools. The video files (and especially unbaited camera footage) provide substantial repositories of information awaiting future investigations, the value of which may not be fully appreciated until remote ecosystems such as the Pilbara rivers are changed by human activities and the inevitable invasion of alien species occurs.

SURVEY STRATEGIES

In remote areas where fish surveys are often focussed on maximizing gamma diversity estimates (obtaining species richness at the catchment or among waterhole scale), rapid survey of the maximum number of sites is often desirable. Where small water holes are encountered and habitat complexity is minimal, estimation of species richness can often be achieved within a few hours of netting (*e.g*. beach seining and gillnetting), but capture and handling can affect survival of fishes. Large waterholes present a different logistical issue, however, and the time spent at these sites has to be traded-off with transit time between waterholes, and ultimately affects how many sites can be visited. As yet, comprehensive data do not exist for retrospectively determining best strategies for estimating gamma diversity in dry-land rivers at the catchment scale.

Data obtained in this study afford an initial, albeit preliminary, insight into strategy for surveying dry-land river catchments based on video- and netting-based techniques. To this end, a range of strategies have been selected, ordered according to time requirements in the field and empirical data used to give some indication of the trade-offs between information gained in relation to effort (Table VI). In terms of calculating alpha diversity (species richness in each waterhole), netting strategies performed relatively poorly (strategies 2 and 5), whereas camera strategies (strategies 1 and 3) or a mixed camera and netting strategy (strategy 4) produced estimates closer to those from all techniques combined (strategy 6) (Table VI). If the intention is to comprehensively survey each large waterhole in a study, then strategy 6 is best (Table VI). It is also possible, however, that species remained undetected in waterholes that were surveyed in this study, implying that further effort or inclusion of additional survey techniques may represent an even better strategy.

Beesley & Gilmour (2008) used rotenone to conduct a complete census of small waterholes (surface area: 315–675 m^2) and thus overcame this issue. This was not, however, logistically possible (and is ethically questionable) when scaled-up to much larger waterholes (Table I). From an ecological perspective, large waterholes appear to serve as important refugia for long-term persistence of fish communities in remote dry-land rivers (Arthington *et al*., 2005; Beesley & Prince, 2010). Therefore, it may be wise to conduct further studies where species accumulation and effort relationship are explored with substantially greater survey effort than was used here. The versatility of cameras should prove useful in the process, and follows a precedent in the riverine fish ecology literature, whereby species richness estimates have been assessed as a function of sampling effort based on single techniques (Lyons, 1992; Patton *et al*., 2000; Kennard *et al*., 2006; Ebner *et al*., 2008). The effects of multiple technique combinations on species richness estimates in fish ecology have been afforded far less attention. This study has to some extent been successful in redressing this issue, but clearly a comprehensive understanding of the effects remains some way off. Widespread testing of remote underwater video in clear-water freshwater ecosystems is recommended.

In conclusion, camera-based techniques provided versatile tools for surveying fishes in large waterholes with heterogeneous habitat under high visibility conditions. Subtle differences in species richness estimates resulted from employing of baited and unbaited cameras in this study; however, the depth that cameras were deployed to was a more important driver of species richness estimates. No single survey technique consistently outperformed all of the other techniques in the comparison, and a combination of netting and camera-based techniques maximized species richness estimation in each waterhole. Cameras warrant application in aquatic areas of high conservation value including those where threatened species are the focus of monitoring, or if such species would otherwise be encountered as by-catch by destructive techniques. It is foreseeable that in particularly sensitive cases cameras will provide an invaluable means of studying fishes by informative and non-destructive means.

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