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Application, development and opportunities of Remote Underwater Video for freshwater fisheries management

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Abstract – Remote Underwater Video (RUV) is a promising tool for progressing the future of freshwater fisheries monitoring and management. While uses have previously been focused on marine systems there has been a rise in application for freshwaters. Given the potential for coordinated geographical research using RUVs it is essential that standardised methodologies are described and promoted. We therefore conducted a systematic literature review which returned 185 publications that discussed using RUVs in freshwater environments. These publications used RUVs to measure: abundance, species richness, length-frequency, spawning/mating, behaviour, migration, foraging, size, habitat use, species presence and nesting. There were taxonomic and geographic biases in the results, with commercial salmonid fisheries the primary focus and 49% of published research was performed in North and Central America. While some research has investigated best practices, there are numerous gaps including: determining optimal deployment time in different systems/species compositions, determining suitable acclimation time for behavioural analysis and ascertaining the costs and benefits of using bait as an attractant and stereo-camera for photogrammetry. Until these gaps are addressed, we recommend a cautious set of standards for freshwater RUVs deployment which includes using a standard action camera, recording at ≥30 fps with a resolution of 1080p for 60 minutes. This will ensure that data are broadly comparable between studies. Current bottlenecks in methodology uptake relate to data storage, processing time and cost but this may be overcome with the optimisation of computer vision and machine learning. There are broad opportunities to develop RUV application into a powerful tool for freshwater fisheries management, invasive species detection, and ethological observations if standardised and findability, accessibility, interoperability, and reusability (FAIR) workflows are followed.

Keywords: RUV / novel tools / biodiversity monitoring / baited remote underwater video / method standardisation / ethology

1 Introduction

The development of effective and feasible long-term monitoring programmes is crucial to identifying key drivers of large-scale environmental degradation and determining the efficiency of potential restoration (Lindenmayer and Likens, 2010). *In situ*, multi-dimensional observation data achieved through field monitoring can be used to link key processes and biological responses such as community composition, population dynamics, breeding ecology, foraging rates, behaviour and response to stressors at a landscape scale (Block, 2005; Caravaggi *et al.*, 2017; Lindenmayer *et al.*, 2022). Signals of large-scale biological change may only be detected after multiple sampling seasons and these changes may be non-monotonic, for example, boom-bust dynamics of non-native invasive species and native species population

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*Corresponding author: `bsmhar@leeds.ac.uk`; `j.south@leeds.ac.uk` responses (Haubrock *et al.*, 2022; Lindenmayer *et al.*, 2022). Well-designed monitoring programmes thus need to encompass both long-term and landscape-scale processes, which means that they are extremely resource-intensive in terms of both cost and people time (Lindenmayer *et al.*, 2022). During the current rapid rate of environmental change, finding solutions to overcome these challenges is critical for biodiversity managers to answer key ecological questions, determine long-term changes, provide robust evidence to guide management actions and unequivocally demonstrate the benefit of any intervention investment.

Freshwater ecosystems are facing a biodiversity crisis with freshwater vertebrate populations declining twice as fast as terrestrial or marine populations (Tickner *et al.*, 2020). Monitoring programmes in aquatic environments not only face struggles regarding time and cost, but also require specialised equipment and training. Traditional aquatic survey methods (*e.g.*, trawling, gill nets, electrofishing, trapping) tend to be extractive, destructive and have inherent biases which may produce an inaccurate representation of a given population (Cappo *et al.*, 2006; Cooke and Schramm, 2007). In addition, capture methods, such as catch and release, can elicit behavioural changes which impact fitness. An example of this being the nest abandonment behaviour observed by male black bass (*Micropterus* spp.) after catch and release surveys that leads to the total loss of offspring (Hanson *et al.*, 2007). *In situ* snorkel surveys can be completed to reduce negative animal impacts of capture, but these are biased by observer ability, water conditions and fear responses to the observer therefore, extractive methods are used in tandem to maximise reliability (Weyl *et al.*, 2013; Ebner *et al.*, 2015). Environmental DNA (eDNA) approaches are being increasingly utilised and presented as a solution to aquatic ecosystem sampling limitations (Beng and Corlett, 2020). However, molecular analysis is costly and conclusions based on eDNA are currently restricted to detecting the presence/absence of species with available barcodes, and inferences may be spatially confounded in lotic systems due to downstream transport of genetic material (Beng and Corlett, 2020). Even if analysis advances to the point where eDNA surveys can accurately estimate total abundance or biomass, it would not be able to observe the size structure of the populations, which is an important metric for fisheries. It is also noted that neither capture-based methods nor eDNA allow for in-situ behavioural studies.

Camera traps and remote imaging have been extensively used in terrestrial ecosystems as they increase observation likelihood of larger and rare species and remove negative impacts of capture-based methods (Feyrer *et al.*, 2013; Caravaggi *et al.*, 2017; Delisle *et al.*, 2021). Aerial surveys have been used in terrestrial and marine environments to monitor large mammals, fish, and plant stands, by tracking movement and population sizes, but they are limited by weather conditions and to animals or plants that are not hidden beneath water or tree canopies (Kelaher *et al.*, 2019; Camacho *et al.*, 2023). Remote imaging methods remove the risk of sampling in locations that are inaccessible or unsafe (Harvey *et al.*, 2013; Chaudoin *et al.*, 2015). Furthermore, results can be quickly validated by reviewing video data, unlike eDNA and aerial surveys, which often need ground-truthing, and datasets are archived for future reference and analysis, thus making it ideal for long-term monitoring programmes looking to maximise data collection (Hitt *et al.*, 2021).

Table 1. Keyword combinations used in initial literature search.

Keyword combination
"Freshwater"* AND "RUV"*
"Freshwater"* AND "BRUV"*
"Freshwater"* AND "Remote Underwater Video"*
"Freshwater"* AND "Baited Remote Underwater Video"*
"Freshwater"* AND "Underwater Camera"*
"Lake"* AND "Remote Underwater Video"*
"River"* AND "Remote Underwater Video"*
"Stream"* AND "Remote Underwater Video"*
"Reservoir"* AND "Remote Underwater Video"*

In aquatic systems, above-surface cameras have been used to monitor Atlantic salmon (*Salmo salar*) farm escapes, species assemblages, migration patterns and barriers (Shortis and Otis, 2014; Morán-López and Uceda-Tolosa, 2017; Morán-López and Uceda-Tolosa, 2020). Technological advancement, such as waterproof camera housing able to withstand high pressure, has facilitated the application of Remote Underwater Video (RUV) systems. Use of RUV and Baited RUV (BRUV) has been applied broadly in marine systems and is now a common part of the marine fisheries assessment toolkit as they provide fishery-independent data which is efficient, low-cost, and comparable across locations (Mallet and Pelletier, 2014; Whitmarsh *et al.*, 2017). However, the application of remote underwater video in freshwater lags behind that in marine environments despite the potential for innovative monitoring.

The purpose of this review is to synthesise the current literature on the application of RUVs in freshwater and recommend a standardised methodology for effective and comparable monitoring efforts. Where possible we have identified the methods and objectives of freshwater RUV studies and categorised them according to the study objectives. In doing so, we provide a roadmap for using RUVs in freshwater aquatic systems. We also provide a starting point to advance freshwater RUVs' best practices to ensure robust data collection and enhance scientific development while addressing critical knowledge gaps in conservation and fisheries science.

2 Methods

A comprehensive systematic literature search was conducted using the Institute of Scientific Information (ISI; Thomson Reuters) Web of Science online database and Google Scholar database. These databases were searched to find any literature that contained relevant information regarding RUVs in freshwater published up to December 2024. The search term included a range of keyword combinations (Tab. 1). The guidelines for Preferred Reporting Items for Systematic reviews and Meta-Analysis (PRISMA) (Page *et al.*, 2021a; Page *et al.*, 2021b) were used to report this systematic

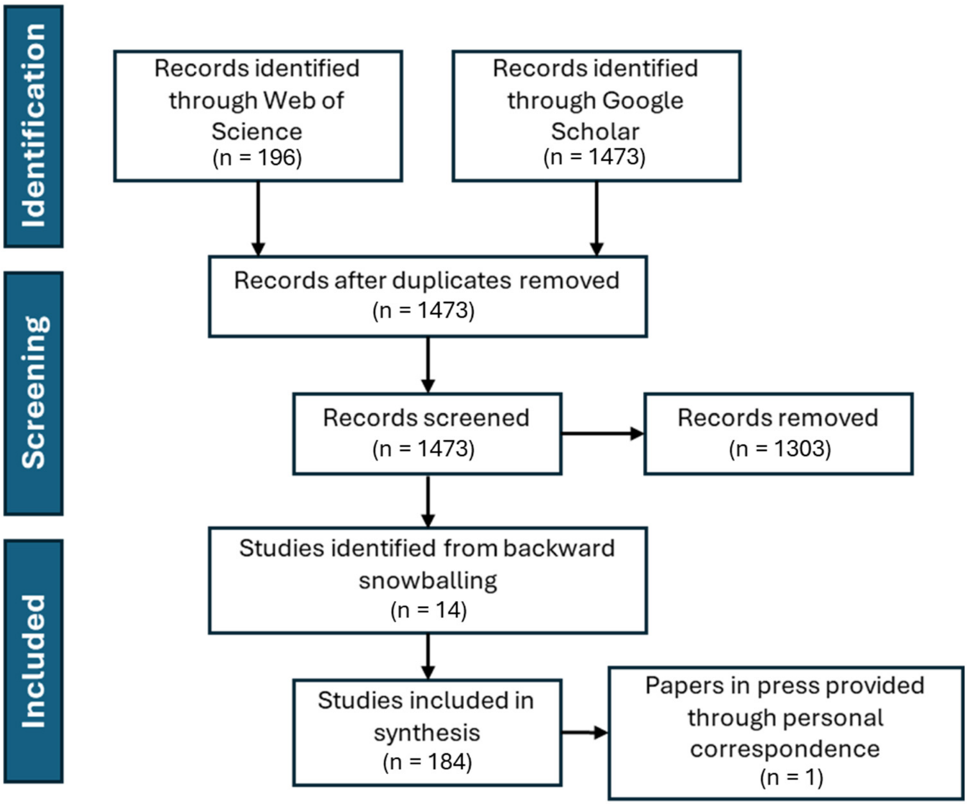

Fig. 1. PRISMA flow diagram illustrating the different phases of the systematic literature review data identification and inclusion.

literature review. The results obtained were collected and assessed by a single reviewer.

Each publication returned was examined and included if it involved using RUVs in a freshwater environment. For each publication, we recorded the following (if details were available); What was being measured/observed in the study, the year of publication, the country the study was undertaken in, the focus species of the study, the waterbody type that the study was conducted in, and the methods used. After reviewing each of the returned pieces of literature, a backward snowball of references was conducted to check for any further relevant literature, which was then incorporated into the database (Wohlin, 2014). Each of these studies were then categorised by the focus of the study. Literature, which was relevant but did not consist of a specific measurement/observation, was also noted and explored further for information that was relevant to this review.

3 Results

A total of 185 unique pieces of literature were identified through database searching, backward snowballing and personal correspondence (Fig. 1). The first published literature on freshwater applications of RUVs was in 1988, which documented swim-up and downstream movement of newly emerged Sea Trout (*Salmo trutta*) fry in the River Itchen, UK (Moore and Scott, 1988). Annual publications were sporadic and in small numbers until 2014 when there was a sharp increase in publication rate (Fig. 2). The database search returned studies undertaken in 27 different countries, spanning all six inhabited continents.

3.1 Survey design

Method specificity and reporting standards are needed for reproducible and comparable RUV research. Despite this, two thirds of the literature did not include information on at least one of these factors (Tab. 2). Prior to 2014, most studies used expensive professional cameras. Since 2014, most RUV studies in freshwater have used some form of action camera. A broad range of frame rates were returned by the literature search (Tab. 2). These ranged from a time lapse video of 1 frame every 5 s to capture habitat use by threatened species (Hannweg *et al.*, 2020a), to a slow-motion video recording at 240 fps trying to capture biomechanics of foraging (Moran *et al.*, 2019). Most studies returned by the literature search did not account for any acclimation time between the initial deployment of the RUV and recording results (Tab. 2).

Remote Underwater Videos with bait arms attached (BRUVs) within the camera's field of view to attract individuals was a common customisation (20% of surveys used some form of baited arm). Bait material and volume was varied, including: bread and marmite, cat food, freshwater fish carcasses and fish eggs (Sup Mat).

The duration of deployments varied greatly. Deployment time is listed as the duration of a single camera deployment, some studies conducted multiple short deployments across multiple microhabitats, whereas others used a single

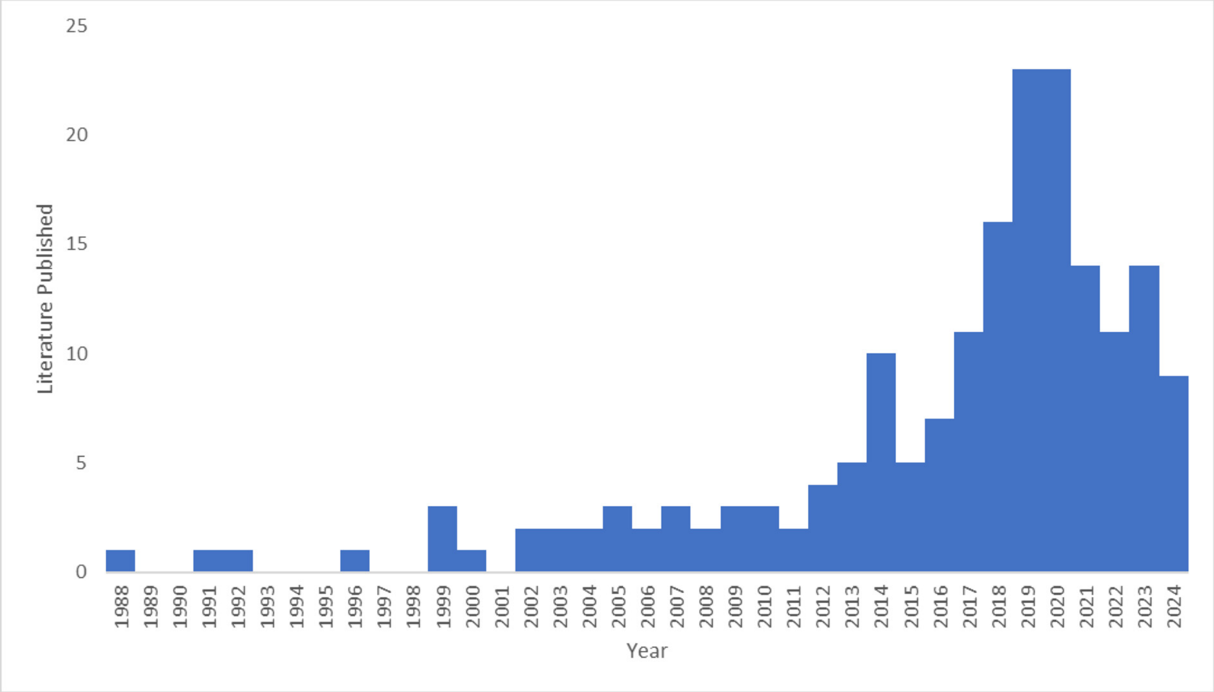

Fig. 2. Histogram of literature related to RUVs in freshwater released each year.

Table 2. Prevalence of technology and methods reporting in the literature.

Factor	Most common method	Percentage of papers using most common method	Percentage of papers that do not specify factor
Camera Type	Action Camera	69%	24%
Frames per second	30 fps	13%	68%
Resolution	1080p	18%	68%
Acclimation Time	0 minutes	11%	64%
Deployment Time (total)	60 minutes	12%	24%
Video Analysis	Human Reviewer	58%	22%

deployment at one site for an extended time. When the RUV is used for a rapid deployment and not left continuously recording the most common duration (12%, $n = 22$) was for 60 minutes and the second most common duration (8%, $n = 14$) was for 30 minutes. Other durations have been used in a more sporadic distribution, with the longest, none-continuous single deployment lasting for 960 minutes (Holubová *et al.*, 2019).

with camera battery life and limited budgets and tight deadlines, and results being required from a single season of surveying, playing a key factor in this.

A range of different video analysis methods were employed throughout the literature. The most common method (58%, $n = 108$) was by a human reviewer watching the footage and manually noting observations. The other method involved the use of specialist software, *e.g.*, EventMeasure (seagis.com.au), Everfocus (everfocus.com), Beast Software (beast.community), Argus (argussoft.org) and Tracker 5.1 (physlets.org) to review footage (13%, $n = 24$). These softwares range in complexity and cost.

3.2 Abundance

Assessing species abundance is a critical component of conservation and wildlife management. This was the most common application of RUVs with 36% ($n = 64$) of studies aiming to determine relative or total abundance of aquatic biota.

The most frequently used method (66%) is calculating the maximum number of individuals, of a single species, observed simultaneously in a single frame known as MaxN (Hitt *et al.*, 2021). This value can be determined by either searching fixed time points (*i.e.*, every 30 seconds) for a frame with the most individuals of a target species recorded (Hannweg *et al.*, 2020b) or obtained by reviewing the entire video (Crook *et al.*, 2021). Alternatively, SumMaxN, a cumulative sum of the MaxN for every species can be used to determine fish abundance within a given habitat or area (Work and Jennings, 2019). If duplicate counting of animals was not considered a confound, *e.g.*, fish passing through a one-way fish pass (Johnson *et al.*, 2007), counting the total number of fish observed (25% of records) was used to estimate species biomass by either estimating the densities and percentage-cover of target species (Karatayev *et al.*, 2021) or using video quadrants (Andres *et al.*, 2020).

3.3 Species richness

Identifying species richness and diversity usually relies on being able to physically handle an animal to key it out correctly or through eDNA analysis, which is compromised in tropical localities by limited barcode libraries and cryptic species. RUVs may offer an alternative to species richness and biodiversity assessments. Overall, 24% ($n = 44$) of studies utilised RUVs for this task with 83% either reviewing the footage in full (Glassman *et al.*, 2022), or by selecting a random frame every minute (Robinson *et al.*, 2019) to identify all individuals to species level and create a species list. Generally, the quality of the video allowed species to be identified to species level by experts and through consulting identification guides for diagnostic characteristics (van Wyk *et al.*, 2017; Pedersen, 2021). In some instances, differentiation between similar species can be confounded due to visibility issues and cryptic taxonomy (Cooke and Schreer, 2002; Widmer *et al.*, 2019). Therefore, identifying organisms to family level occurred in 11% of publications.

3.4 Spawning/mating/nesting

Studying reproductive behaviour of aquatic biota without causing adverse impacts on the target species is fraught with difficulties. RUVs were used in 16% ($n = 30$) of studies where 74% deployed RUVs at known spawning sites to confirm event timings. In some cases, this was used as a solution to better understand spawning of critically endangered fish such as the Devils Hole Pupfish (*Cyprinodon diabolis*) (Chaudoin *et al.*, 2015). Furthermore, RUVs can visually assess effects of abiotic factors such as habitat (Groves and Chandler, 1999) and the lunar cycle (Fernández *et al.*, 2021) on salmonid spawning. RUVs can also be used in tandem with hydrophones to expand the toolkit of remote sensing in aquatic habitats; for example, a novel method of time-synchronised sound and video was also used to identify the sounds produced by spawning trout (Johnson *et al.*, 2018). Opportunistic sampling was able to document footage of courting and mating behaviour incidentally during recording. One of these studies provided the first observation of copulation of the cryptic Andean Catfish (Mena-Valenzuela *et al.*, 2022). Another observation was courtship of Zebrafish (*Danio rerio*) which was recorded outside of the known mating season by chance in a separate RUV study (Sundin *et al.*, 2019). RUVs were positioned directly in front of the nests of target species to observe nesting behaviour. Behaviours ranged from nest guarding and maintenance behaviours conducted by males (Unger *et al.*, 2020), to interactions with heterospecific species eggs placed within the nest (Yamane *et al.*, 2016), subordinates within nests, sheltering with juveniles, and cooperative breeding (Satoh *et al.*, 2022). One study used a less targeted approach where RUVs were deployed in the USA to opportunistically record nesting of Redbreast Sunfish (*Lepomis auratus*) (Martin and Irwin, 2010).

3.5 Behaviour

Behavioural studies comprised 14% ($n = 25$) of the literature, where responses to external stimuli including anthropogenic sounds (Fleissner *et al.*, 2022), chemical cues such as chemical predatory alarm cues (Friesen and Chivers, 2006), infrared lighting (O'Malley *et al.*, 2018) and researchers shadows (Smith, 2022) formed the majority of the analysis. Beyond that, general behavioural time budget research, such as time spent swimming compared to nesting or foraging was the core focus of these papers. RUVs were used to quantify the extent of inter and intraspecific interactions, for example territorial defence behaviours in fish (Ebner *et al.*, 2017), aggressive interactions between noble crayfish (*Astacus astacus*) individuals within the same trap (Raugstad, 2019), and competitive interactions between invasive and native crayfish species (O'Hea Miller *et al.*, 2022a). Collective behaviour of fishes was assessed through open water RUV deployment, where a focal species was identified but data collected on the icthyofaunal community as a whole.

3.6 Migration

Migration studies comprised 10% ($n = 18$), where 56% of this subset deployed RUVs into obstacles passed by migrating species including fish ladders (Negrea *et al.*, 2014), fishways (Limaye, 2019), weirs (Marston, 2014) and fish passes (Hawkins *et al.*, 2018). These RUVs were deployed primarily for commercially important species Alewives (*Alosa pseudoharengus*), Rainbow Trout and Sockeye Salmon (*Oncorhynchus nerka*). The RUVs were left to continuously record, and the footage was reviewed to count the total number of individuals that passed either upstream or downstream. In some cases, *a priori* knowledge of important migratory locations were chosen for RUV deployment whereas in others, RUVs were deployed at waterbody entrances of specialised migration swim-through chutes to monitor salmonid migration (Musslewhite, 2020). Finally, multiple camera arrays of RUVs were strategically placed along the Atlantic Salmon migration route to incorporate the spatio-temporal aspect of migrating populations (Borgstrøm *et al.*, 2010).

3.7 Foraging

Foraging activities comprised 7% ($n = 13$) of the overall results. These recordings provided information on trophic interactions to quantify intensity of bottom foraging (Pledger *et al.*, 2014), feeding aggregations (Starrs *et al.*, 2015) and species feeding on floating material including zooplankton (Marchand *et al.*, 2002) and recently released eggs (Šmejkal *et al.*, 2017). Baited studies comprised 31% ($n = 4$) with the intention of observing scavenger feeding behaviour with the most common bait being a fish carcass positioned in front of the RUV (Unger and Hickman, 2019). High frame rate video was used to record predation events by Bluegill (*Lepomis macrochirus*) on live prey tethered within view of the RUV (Moran *et al.*, 2019).

3.8 Size

Estimating length underwater of a moving object is a barrier to ascertaining critical fisheries data, thus length frequency studies made up 7% ($n = 13$) of the total results. The most common method (62%) estimated size by comparing an individual to an object of known size that is within the frame of the video. These objects were variable and included a stake of known size (Tweedie *et al.*, 2018), to specially mounted scale bars (Loffredo, 2018). In some instances, exact sizes were not calculated, instead individuals were either assigned a size class bin (Hopper, 2019) or estimated in relation to previously caught individuals (Skorulis *et al.*, 2021). Specially calibrated stereo-RUVs and specialist software was used in 18% of studies to accurately estimate the size of individuals. One novel method was applied in Canada to study Shortnose Sturgeon (*Acipenser brevirostrum*), where parallel lasers were mounted to an RUV and used to estimate length (Usvyatsov *et al.*, 2012). Another novel method involved counting the pixels a fish took up in footage when it passed a known location and transforming these to millimetres to estimate sizes, as done in Reunion Islands to estimate the size of Red-tailed Goby (*Sicyopterus lagocephalus*) (Boussarie *et al.*, 2016).

3.9 Habitat use

RUVs were used to assess habitat associations in 7% ($n = 13$) of the results. Most studies aimed to determine differences in artificial habitat use compared to natural habitat as well as assessment of seasonal habitat changes (Pratt *et al.*, 2005; Lintermans *et al.*, 2013). RUVs were also used to quantify cyprinid habitat use after disturbance events from hydropeaking (Boavida *et al.*, 2021).

3.10 Presence

A small number of studies (5%, $n = 10$) were intended to confirm species presence in a waterbody. Most studies deployed RUVs at a fixed location within the waterbody in the hope of serendipitously detecting a species, such as the first official record of the Cleft-lipped Goby (*Sicyopterus cynocephalus*) in Australia and multiple deployments were used to determine the presence of escaped farmed salmon in Norway (Ebner *et al.*, 2017; Svenning *et al.*, 2017). Macrophyte studies were rare across the dataset but short RUV deployments were completed in multiple locations in the UK to confirm the maximum colonisation depths of all macrophytes in the waterbody (Spears *et al.*, 2009).

4 Case study applications for fisheries science and conservation

4.1 Conservation intervention

Eradication of non-native invasive species is a high-risk, high-cost venture which relies on robust evidence-gathering and success post-intervention. A successful example of an invasive fish eradication project took place at the Rondegat River in the Cape Fold Ecoregion, South Africa (Marr *et al.*, 2012; Weyl *et al.*, 2013; Weyl *et al.*, 2014; Weyl *et al.*, 2016), an important biodiversity hotspot characterised by high diversity and endemism (Ellender *et al.*, 2017; Broom *et al.*, 2023). RUVs were deployed yearly to effectively monitor the recovery of the endemic fish population in a non-destructive manner (Weyl *et al.*, 2013; Weyl *et al.*, 2014; Weyl *et al.*, 2016). Using the Rondegat River's RUV yearly dataset (2011–2016), Castañeda *et al.* (2020) tracked the occupancy dynamics of the endemic fishes along the river, before and after the eradication of the invasive fish. They found that the strongest driver of the endemic fish's probability of occupancy in the river was the presence of an invasive fish. After the invasive fish eradication, the endemic fish were able to naturally colonise downstream sections of the river and increase in density. Two of the endemic fish populations appear to have reached population equilibrium across the river, while the third has not, suggesting it may be more sensitive to fluctuations in habitat variables (Castañeda *et al.*, 2020). To assess the habitat associations of the three vulnerable and recovering cyprinid species, Broom *et al.* (2022) utilised a RUV system across 51 sites as part of a long-term monitoring project for the Rondegat River post-intervention. With repeated sampling over three seasons (2018–2019), Broom *et al.* (2022) were able to assess community composition and relative abundances with respect to habitat, its overlap with a protected area and species-specific abiotic predictors of relative abundance. Results confirmed a lack of re-invasion and indicated that habitat-specific interventions to reduce the impact of drought, eutrophication and sand deposition are needed along the Rondegat River to ensure the continued persistence of threatened fish. Underwater video monitoring is an effective and low-cost approach that can rapidly inform tangible conservation recommendations for vulnerable fish species in impacted or recovering river systems, especially in locations with underfunded resourcing for biodiversity management.

4.2 Ethology

O'Hea Miller *et al.* (2022a) investigated the competitive interactions and outcomes between an invasive crayfish (*Cherax destructor*) and a critically endangered native one (*Euastacus dharawalus*), which cohabit a 7.5 km stretch of creek in Wildes Meadow, located in the Southern Highlands region of NSW, Australia. Up until this point, behavioural investigations of invasive and native crayfish had largely been confined to laboratory trials, which facilitate clear and controlled observations of individuals competing over resources (Cerato *et al.*, 2019; Lopez *et al.*, 2019; O'Hea Miller *et al.*, 2022b). Owing to the challenges of observing individuals in situ, particularly due to shallow and often turbid conditions, little was known about how these species interacted under natural conditions nor whether body size affected contest dynamics and outcome. By deploying 15 baited remote underwater videos along nine locations within the creek over the course of 12 months, O'Hea Miller *et al.* (2022a) were able to extract and score 178 interspecific and intraspecific interactions from which interaction duration, maximum intensity, conclusion, outcome, and interaction initiator were quantified. All behaviours were assessed relative to an established ethogram (Bergman and Moore, 2003) and relative body size of contestants was estimated from the precent difference in size as measured on the video screen (Martin and Moore, 2007). Overall, *Euastacus dharawalus* won more contests than the invasive species *Cherax destructor*; however this was largely attributed to the fact that in most cases, *E. dharawalus* was larger than *C. destructor*. Alarmingly, when considering only interactions where contestants were size-matched (*i.e.*, within 10% body size), *C. destructor* was more likely to win interactions than *E. dharawalus*. Additionally, *C. destructor* were more willing to initiate contests than *E. dharawalus*, even if *C. destructor* was the smaller contestant, and they were more willing to continue fighting than *E. dharawalus* in intraspecific contests, demonstrating a greater inherent aggressiveness of the invasive species. This study highlights the capacity of BRUVs to quantify complex behavioural interactions in challenging freshwater systems, but also a key consideration of using BRUVs for behavioural studies – namely the potential for limited and unbalanced sample sizes. Firstly, some BRUV deployments had to be discarded from the analyses due to high turbidity and time of year (*i.e.*, winter crayfish inactivity). Secondly, O'Hea Miller *et al.* (2022a) reported only one interaction where *E. dharawalus* was smaller than *C. destructor* (compared to 29 interactions where *C. destructor* was smaller than *E. dharawalus*). Future considerations must, therefore, involve increasing the number of BRUV deployments over time and the use of stereo-BRUVs to enable scoring of relative body sizes of all individuals in the frame. Both these considerations will help boost replication of behavioural observations and hence the efficacy of BRUVs for understanding behaviour in freshwater systems. Incorporating ethological studies and interspecific interactions into fisheries management plans has been a persistent and key challenge which may be tackled by appropriate use and deployment of BRUV systems.

4.3 Fisheries monitoring

Chambo (*Oreochromis* spp.) is a key fishery in Lake Malawi and is a key target for management and conservation efforts. However, traditional monitoring methods for Chambo, such as gillnetting and trawling, are destructive and can negatively impact fish populations (Tweddle and Magasa, 1989; Banda *et al.*, 2005; Weyl, 2005; Weyl *et al.*, 2010).

Van Wyk (2019) evaluated the potential for using stereo-BRUVs to monitor Chambo populations across different management zones of Lake Malawi (Mozambique and Malawi) to determine the optimal sampling design for annual monitoring. Both Chambo abundance and size differed significantly between Malawi and Mozambique - which may be attributed to differences in fisheries pressure. Malawi experiences greater levels of fishing pressure compared to Mozambique, resulting in a decline in Chambo populations and a decrease in the size of sexually mature individuals. In contrast, Mozambique has relatively low fishing pressure due to low population densities, weak market forces, and a history of civil war. This has resulted in higher Chambo abundance and larger sexually mature individuals in Mozambique compared to Malawi (Halafo *et al.*, 2004; Weyl, 2005; Weyl *et al.*, 2010; van Wyk, 2019). In this system, an acceptable stereo-BRUVs deployment time was 15 minutes and required a maximum of 120 annual video samples to detect a 10% change in Chambo abundance over a hypothetical 10-year monitoring scenario. This suggests that stereo-BRUVs can be used as a cost-effective long-term monitoring tool for economically and ecologically important fisheries, provide evidence-based recommendations for the establishment of closed sanctuary areas, and monitor intervention outcomes.

This case study highlights the importance of effective, standardisable monitoring methods for fisheries management and conservation, and the potential of stereo-BRUVs technology to provide robust data for monitoring and managing complex inland fisheries.

5 Discussion

Despite RUV research being completed across a broad range of subject matter and spanning continents, there is a distinct lack of cohesion in method standardisation between research groups. This is a stark contrast to the marine environment, where proven methodological standards are in place and coordination levels at a global scale are high. The lack of standardisation within the freshwater environment limits the potential comparisons between studies or meta-analyses, thus hampering RUV work being used to its full capacity in freshwater fisheries. Research so far has, therefore, been fairly ad-hoc, with a sharp increase in the literature published annually from 2014, which likely reflects a technological trend in better video quality, combined with a decrease in the cost of cameras, which has made action cameras more available. The drop in frequency of literature using freshwater RUVs between 2021 and 2022 is believed to be a result of COVID-19 lag in publishing, with limited studies occurring during the pandemic lockdowns. However, there does not appear to have been a rebound in the volume of annual publications and the number of publications each year post 2020 have remained stable. Several studies have used historical data that is older than 5 years, while others do not specify the timeframe of data collection. Analysis protocols, such as human reviewers watching 100+ hours of footage to obtain a complete species list, will be more time consuming than video analysis that automatically identifies the presence of an individuals through Artificial Intelligence. Overall, the processing delay between data collection and publication is entirely reliant on the resources available and the questions that are being asked.

Table 3. Recommended standards and reasoning.

Factor	Recommendation	Why?
Camera Type	Action Camera	Most common in reviewed literature
Frames per second	30 fps lower limit	To reduce blur produced by fast moving individuals
Resolution	1080p lower limit	Suitable resolution to identify species while saving on storage space
FOV	109° and 120°	Creates a suitable visible area without distorting the image
Acclimation Time	1 minute	Majority of literature does not have an acclimation time incorporated, but to avoid issues with disturbed sediment obscuring field of view, an acclimation time is recommended
Deployment Time (total)	60 minutes	Most common in reviewed literature and standard in marine systems. Over collection of data is preferred over under sampling until sampling efficiency is assessed systematically in freshwater
Video Analysis	Human Reviewer	Most accessible option in terms of cost

5.1 Use of RUVs

Freshwater research utilising RUVs has the capacity to be a one size fits all method for ecological assessments and fisheries science, if deployed correctly and in a standardised manner. Development of a freshwater RUV consortium following a standardised methodology could result in globally coordinated research which spans broad spatio-temporal ranges with the capacity to answer pressing questions in fisheries science. Similar consortiums have been created, for example: acoustic tracking in marine systems and BRUV census of global shark populations, which have resulted in unexpected natural history observations (Phillips *et al.*, 2019; MacNeil *et al.*, 2020; Lennox *et al.*, 2024). As RUVs represent a long-term low-cost data acquisition method they are an excellent tool for post conservation intervention monitoring, as once the initial costs of purchasing the equipment are covered, future costs are limited to staff cost, travel cost and data storage and processing (Harwood *et al.*, 2025). They can also be used to complete rapid baseline assessments of freshwater environments, perform freshwater fisheries stock assessments or monitor the escapements of farmed fish (Weyl *et al.*, 2013; Svenning *et al.*, 2017). This is especially true for stereo-B/RUVs which provide accurate length estimates for length frequency and biomass estimations which can be used to support fisheries independent assessments.

5.2 Limitations of RUVs

With most modern action cameras maximum battery life is around 80 minutes, however, this can be lower in colder waters, meaning that RUVs with action cameras cannot be deployed for extended periods of time, which can result in key events being missed. Other issues include malfunctioning memory cards, water damage, unfocused images and obstacles within the field of view, all of which can result in wasted effort (Struthers, 2015). Although depending on the specific set up of the rig, *i.e.*, commercial or home-made, there may be low-tech ways to mitigate these, with low-cost extended battery life camera system being developed (Magneville *et al.*, 2023; Fetterplace *et al.*, 2023; Dunkley *et al.*, 2023). There are also environmental limitations into RUV surveys, such as light levels and turbidity limiting data capture. Turbid water greatly reduces an RUVs field of view which reduces probability of event observation. Similarly suitable lighting and hours of daylight restricts timing of many surveys and introduces bias against nocturnal species - which may also avoid artificial lighting (Struthers *et al.*, 2015). This highlights a priority question to advance methodology by assessing the effectiveness of white, red and blue lights for RUV surveys.

5.3 Camera setup and analysis

A set of standards for using stereo-BRUVs in marine environments has been proposed (Langlois *et al.*, 2020) and these have been used as a framework to guide the standards described here, to ensure reliable deployments of all forms of RUVs in freshwater. To be able to create a standardised method for RUVs in fisheries research there are several systematic methodological developments that need to be considered (Tab. 3).

Any action camera which can record at the recommended settings can be used for future standardised surveys. Framerate, resolution and field of view (FOV) are all important considerations as they influence a video analyst's ability to accurately identify, count and measure fish, as well as the size of the visible area (in the case of FOV). At a minimum, high definition (1920 × 1080p) resolution with 30 frames per second, with a field of view between 109° and 120°, appear to be an adequate standard (Langlois *et al.*, 2020). While 4K resolutions might be tempting, researchers will run into challenges with cameras overheating and space for data storage. For measuring fish length with stereo-camera any settings that automatically adjust the pixel size (*e.g.*, image stabilisation), frame rate (*e.g.*, auto low light) or distort the image (*e.g.*, fish-eye or ultra-wide FOV) should be disabled (Langlois *et al.*, 2020). Standardisation of survey areas can be achieved through the placement of quadrats of known size on the bed of the waterbody, within the camera's field of view (Longo *et al.*, 2018).

Acclimation time is likely to vary between species and communities depending on their exposure to disturbance and life history traits. Most studies reviewed here did not include an acclimation time as most species entered the RUV's field of view a short period of time after deployment. This is a methodology priority question - to determine a suitable baseline acclimation time for different species and purposes to ascertain whether a standard can be achieved. Similarly, a standard operating procedure for total deployment time has not yet been determined, thus we recommend the most common deployment duration time, *i.e.*, 60 minutes. However, van Wyk (2019) found that 15 minutes was sufficient for fisheries monitoring in Lake Malawi, whereas 60 and 30 minutes is a recommended deployment time in marine systems to reduce diminishing returns (Langlois *et al.*, 2020). Sampling efficiency analyses such as time - species accumulation curves in freshwaters are required urgently.

Video analysis by Artificial Intelligence is in development, this AI would be able to identify frames of the footage that hold a target species that can then be analysed by a human reviewer. Artificial Intelligence has been used for the automated identification of marine fish and invertebrates (Bürgi *et al.*, 2025), but large training datasets are required in order to achieve these results (Ditria *et al.*, 2020). For Artificial Intelligence to be successful in freshwater environments these large training datasets are an area for future work. Potentially, in the future there is even the scope that it can fully review footage identifying all species and individuals that occur throughout a video sample, but prior to optimisation and validation of these models human review should be prioritised. Ideally, reviewers will have undergone species identification and software training prior to analysis. Random review should be implemented for quality assurance, and if a complex community is present then two independent reviewers should be used to ensure accuracy. While human review is currently the only viable option, researchers should invest effort into developing training datasets to enable AI applications when the technology is mature.

Compared to RUVs, BRUVs achieve higher MaxN and species richness estimates. Therefore, BRUVs should be used for abundance, species richness and presence absence studies. Due to bait-attraction altering natural behaviour we do not advise their use for ethological observations or habitat use, nesting and migration. On the other hand, BRUVs can be used effectively for some ethology experimental purposes such as scoring competitive behaviours and aggression (O'Hea Miller *et al.*, 2022a).

Experimental bait efficiency assessments need to be completed to recommend a data-driven standard, where bait type, volume, local hydrology and survey purpose must be considered. Using a bait local to the area is recommended for practical purposes, small oily food fish species work well as the scent plume travels well in the water, and they are usually inexpensive and readily available. The use of local fish reduces the risk of the introduction of disease or invasive species while conducting surveys. Nonetheless, non-natural bait has also been used, *e.g.*, marmite^{TM} and bread (Bajaba *et al.*, 2021), variations of this, or indeed canned oily fish may be preferable if surveying in remote locations. The use of marine species as bait in freshwater has been promoted in the UK during crayfish surveys to reduce chances of disease introduction through freshwater bait (J South pers. com.). Furthermore, depending on survey location, safety concerns should also be considered when using BRUVs as bait has been known to attract large predators, like crocodiles, that pose a risk to researchers and BRUVs should be avoided when these risks are present (King *et al.*, 2018). Effort should be applied to follow FAIR (Findable, Accessible, Interoperable and Reusable) data workflows to the large amount of data produced in RUV surveys (de Visser *et al.*, 2023). This study has highlighted that more than 10% of all studies are missing details about protocols, illustrating that more needs to be done to meet the FAIR data workflow requirements for detailed metadata about protocols.

As standard, all data should be suitably annotated with meta-data for location, date and surveyor, and saved in raw video format. Good practices regarding data management and storage are crucial and data ought to be stored along with off-site back-ups in both physical and cloud repositories. However, this may incur unforeseen costs for practitioners and researchers. We strongly recommend the creation of a global freshwater RUV repository, following open data principles, similar to those suggested for marine systems (Langlois *et al.*, 2020). Standard approaches to analysis will enhance the usability and interoperability of datasets and analysis codes. All species should be identified to the lowest taxonomic level possible. MaxN analysis methods should be used as standard in abundance studies, as it is less likely to overestimate true abundance (van Wyk, 2019). Efficiency and robustness of MaxN calculation approaches, such as snapshots vs total video, need to be assessed to recommend best practice. MaxN may underestimate abundances if there are instances of several groups of individuals, newly proposed Synchronised MaxN (SMaxN) could offer a more accurate estimate when videos include several groups of individuals (Magneville *et al.*, 2024). Specialist software, such as EventMeasure, can be used to annotate video which can ensure that fish are not counted multiple times or missed when reviewing footage to reduce the risk of miscounts when calculating MaxN values. This software can also be used with stereo-RUVs to obtain accurate fish lengths after careful calibration (Langlois *et al.*, 2020). When possible, specialist software should be used to ensure that results are consistent; however, the high costs of licences for this software often makes it implausible. Regardless of the use of specialist software, the suggested standard methods should be used when reviewing footage, with footage saved for future review if requested. Behavioural studies should follow published ethograms when possible, however, the reviewing process can be lengthy and subject to observer bias. Using free software such as BORIS (Friard and Gamba, 2016) the recommended standard method of analysis would be to review footage back at an increased speed, until a desired event is observed, and then reviewing the footage at normal speed to score results, as in O'Hea Miller *et al.* (2022a). Automated behavioural analysis software is available, but the cost is often prohibitive for environmental managers and negates the initial RUV cost saving.

5.4 Future steps and potential developments

The prospect of low cost, high data acquisition methods for fisheries monitoring and management means that technology is constantly developing for both research and commercial applications. Novel methods including a Raspberry-Pi platform can allow automated data acquisition through scheduling and automatically uploading results to an online database (Almero *et al.*, 2021), or through a video streaming link that listens for requests to connect (Dadios *et al.*, 2022), removing the need for researchers to replace batteries and storage cards. This technology is in the initial trial stages and currently the video is limited to 6 fps, which does not adhere to the standards suggested.

Time spent reviewing footage is a major bottleneck in RUV methodology, this may be overcome with optimisation of artificial intelligence applications and machine learning to automate the process. Through deep-learning, AI can be trained to categorise behaviours and identify fish species with the most recent versions being able to detect fish and categorise species to almost a human-like accuracy (Abangan *et al.*, 2023). Approaches such as Convolutional Neural Networks or the You Only Look Once (YOLO) algorithm can be optimised to identify species passing through fish passes or in RUV footage to automatically identify species (Ovalle *et al.*, 2022; Soom *et al.*, 2022). For instances when species identification is not plausible AI, could be used instead to flag instances when individuals are present on the screen so that a human observer can manually review a smaller subsection of the video file with confirmed presence, rather than reviewing footage without species present. Furthermore, this could be developed further to enable the monitoring and tracking of individual fish over time, calculate rapid biomass estimated by deriving length-weight relationships, and expedite the accurate detection of invasive fish through catchments, thus potentially revolutionising the way we monitor and manage aquatic ecosystems. Environmental factors including turbidity and lighting limit the effectiveness of RUVs, specialist lighting rigs can be developed to address lighting issues; for example, a clear liquid optical chamber to improve underwater visibility (Jones *et al.*, 2019).

Remote Operated Vehicles are a move away from the static camera approach and should be considered a separate methodology entirely, with method development focused on in their own right. These have almost exclusively been used in marine environments but are increasingly being exploited in commercial applications in freshwater. For example, ROVs have been deployed effectively in reservoirs to assess the presence and distribution of target species, such as Signal Crayfish (*Pacifastacus leniusculus*) (P.D. Stebbing pers com.). Information gathered from such surveys has facilitated determining the risk of spread of invasive non-native species from reservoir assets and their distribution within the asset providing valuable information for the development of biosecurity and management plans. Additionally, ROVs are able to detect signs of crayfish, such as burrows, burrow bound animals, and parts of animals, such as claws and carapaces, which trapping or static video may miss. This provides much more detailed information on the size and distribution of the population, in addition to key information on meta-population distribution, which is vital in the development of management plans. The deployment of traditional monitoring methods, such as trapping for crayfish, are not suitable for assessing key locations in reservoirs which are often hard and dangerous to access. Draw off towers and scour values present key points of risk for the potential dispersal of invasive non-native species from impounded reservoirs but are difficult to monitor due to health and safety risks presented by the infrastructure and its operation, in addition to the depth of water in which they are often situated. As ROVs can be deployed at a distance and at depth, with umbilical cords of 100 m being common, these issues are overcome. The large size and weight of currently available ROVs does not make them an ideal tool for monitoring smaller freshwater environments. While ROVs are becoming smaller and more accessible, they are preferable in still water or without their umbilical cords to avoid becoming entangled on submerged objects. The biggest issue currently faced by ROVs is that they cannot handle strong water movement, which prevents standardised sampling protocols being followed. Both RUV and ROV, as well as the field of research in freshwater ecosystems, would benefit from method comparison studies to better understand the pros and cons and how these technologies could be used together to provide more holistic ecosystem assessments.

6 Conclusion

RUVs offer a non-destructive and effective method for monitoring freshwater fisheries species in non-turbid waters. They can provide us with very useful information to address a range of scientific questions. All future RUV surveys should consist of an action camera, set to record at 30 fps, 1080p being deployed for 60 minutes. By having a consistent methodology all future surveys can be accurately compared. These standards that we have recommended will ensure that RUV becomes a vital tool in the future of freshwater surveys. Rapid technological advances have the potential to vastly transform fisheries research to become streamlined, automated, and standardised which will improve both the quality and granularity of data that environmental managers have access to. This can greatly advance the robustness of management plans and capacity for evidence-based interventions. With the declining state of global freshwater fisheries and lack of management incentive we promote the creation of an international freshwater RUV consortium to increase standardisation, collaboration and method development to improve data availability and implement baseline monitoring programmes (Barbarossa *et al.*, 2021; Ainsworth *et al.*, 2023).

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Supplementary material

Appendices A. RUV Abundance Studies.
Appendices B. RUV Species Richness Studies.
Appendices C. RUV Spawning/Mating Studies.
Appendices D. RUV Behaviour Studies.
Appendices E. RUV Migration Studies.
Appendices F. RUV Foraging Studies.
Appendices G. RUV Size Studies.
Appendices H. RUV Habitat Use Studies.
Appendices I. RUV Presence Studies.
Appendices J. RUV Nesting Studies.

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