LIMNOLOGY
and
OCEANOGRAPHY: METHODS

Limnol. Oceanogr.: Methods 2019

doi: 10.1002/lom3.10331

Improving visual biodiversity assessments of motile fauna in turbid aquatic environments

Robyn E. Jones,[1]* **Ross A. Griffin**,[2] **Sam C. Rees**,[3] **Richard K.F. Unsworth**[1,3]
[1]College of Science, Swansea University, Swansea, United Kingdom
[2]Ocean Ecology Limited, Epney, United Kingdom
[3]SEACAMS 2, Swansea University, Swansea, United Kingdom

Abstract

Current knowledge of turbid coastlines relies heavily on extractive sampling methods with less destructive visual techniques limited primarily by underwater visibility. Baited Remote Underwater Video (BRUV) is now a commonly used nonextractive sampling technique which involves the use of bait to attract motile fauna to the field of view of the camera, but its use is restricted to clear water environments. Here, we describe and test the addition of a clear liquid optical chamber (CLOC) to a BRUV system to improve underwater visibility when observing motile fauna in turbid waters. The CLOC method was trialed with respect to the ability of the system to identify taxa to species level in both controlled laboratory and field conditions across gradients of underwater visibility. This study found that the introduction of a CLOC to a conventional BRUV system significantly improved the ability to observe identifying features of four fish species in a controlled low-visibility environment ($p \leq 0.001$). The ability to identify taxa to species level in field conditions was also significantly increased with the addition of a CLOC ($p \leq 0.01$). We conclude that the introduction of a CLOC to a conventional BRUV system is a reliable way of improving underwater visibility when assessing motile fauna allowing for a more consistent identification of taxa to species level. This system may be applied to both marine and freshwater aquatic environments.

Turbid coastal waters occur through particles suspended or dissolved in water and are found globally from the tropics to the poles. These particles may include sediments, organic and inorganic matter, algae, and other microscopic organisms (Wilber and Clarke 2001; Smith 2003). The ecological knowledge we have of these environments currently relies heavily on the use of extractive sampling methods limiting their capacity to directly observe either the habitat or associated flora and fauna and is increasingly prohibited in areas covered by Marine Protected Area management. Such areas are commonly considered critically important for biodiversity, fisheries, energy, and increasingly ecosystem services such as carbon storage (Meybeck 1993; Levin et al. 2001). These turbid environments are often areas of the world characterized by rapid population expansion, resource exploitation, and energy development due to their dynamic nature (Mélin and Vantrepotte 2015).

Marine renewable energy development in sectors such as offshore wind, tidal, and wave energy is an example of resource exploitation in these dynamic areas and is considered a key objective in many countries (Inger et al. 2009). Faunal assemblages associated with these developments are currently poorly understood due to the challenges of sampling these environments. Extractive sampling techniques such as trawling and benthic grabbing are usually restricted in their proximity to sensitive habitats as well as seabed infrastructure due to risks of snagging or damage to either the environment, installation, or sampling equipment (Davies et al. 2001; Det Norske Veritas 2010). This therefore reduces the reliability and accuracy of data if methods are implemented at a distance from a target area or installation (Det Norske Veritas 2010; Unsworth et al. 2014; Lindholm et al. 2015; Griffin et al. 2016). Comprehensive baseline assessments and monitoring of coastal biodiversity is essential in light of increasing coastal developments, with these coastal areas considered the interface between the human population and the ocean (Pelc and Fujita 2002; Gill 2005; Sheehan et al. 2010; Heiskanen et al. 2016).

Nondestructive sampling methods such as underwater cameras and other visual survey techniques are currently limited in extreme turbid environments. Such methods rely heavily on good levels of underwater visibility which reduces their reliability when assessing associated biological communities in turbid areas (Davies et al. 2001; Mallet and Pelletier 2014). Acoustic methods can be used as an alternative in this instance; however, these

*Correspondence: rjones7817@gmail.com

techniques are also susceptible to backscatter in areas of high turbulence (Evans and Thomas 2011) and are often costly.

Baited Remote Underwater Video systems (BRUVs) are a suite of techniques which have previously been applied to coastal habitats globally. These methods involve the use of bait to attract motile fauna into the field of view of a camera (Cappo et al. 2006; Mallet and Pelletier 2014). These techniques have primarily been tried and tested in high visibility and biodiverse environments such as those found in Australia and New Zealand (Watson et al. 2005; Cappo et al. 2006; Whitmarsh et al. 2017) but have also been successfully used in the Northern Hemisphere (Griffin et al. 2016). Examples of such are their use in assessing the size and relative abundance of mobile fauna found in temperate coastal seagrass and kelp habitats (Unsworth et al., 2014), and monitoring motile fauna around offshore wind turbines (Griffin et al. 2016). Even within these successful trials, underwater visibility is still identified as a limiting factor in these often turbid waters, with the ability to identify faunal taxa to species level often greatly reduced (Davies et al. 2001; Mallet and Pelletier 2014; Bicknell et al. 2016). For instance, the ability to confidently assess certain families such as Gadidae to species level in low-visibility environments may prove difficult if features including barbel, jaw and fin characteristics are difficult to determine. Further research is therefore required to help expand the working window for using baited cameras as a means of assessing motile fauna in coastal areas where visibility is reduced.

Here, we describe and test a clear liquid optical chamber (CLOC) to improve underwater visibility when observing motile fauna in turbid waters. Such methods have previously been applied to drop-down camera technology to assess benthic habitats in turbid conditions, but their actual effectiveness in improving image clarity and species/habitat identification has not been tested. We describe and expand this use of the CLOC as a form of BRUV system and test this in both controlled and field conditions. This research aimed to test whether employing a CLOC-BRUV system in low-visibility conditions improved image clarity and increased species level identification relative to traditional BRUV systems. For the purpose of this research, the term "motile fauna" refers to fish assemblages and benthic macrofauna likely to be monitored using BRUV methods.

Methods and materials

Comparisons between two remote BRUV camera systems, one equipped with a CLOC and one without were undertaken over a gradient of increasing turbidity in both controlled laboratory conditions and field conditions. An existing stereo-BRUV system (designed to allow stereo vision of a faunal community for measuring) was used to collect the video footage without the presence of a CLOC. For consistency, the footage recorded from only one of the stereo-BRUV cameras (left) was analyzed for this research. These two camera systems were deemed as "remote" as they are free standing on the seabed without the need for an operator (Cappo et al. 2004; Watson et al. 2005).

The CLOC-BRUV system

A custom built frustum stainless-steel frame of the dimensions L170 cm (diagonal length) × W64 cm × H93 cm designed by Ocean Ecology and fabricated by R. W. Davis & Son (Gloucester, UK) (Fig. 1) was used for the CLOC-BRUV deployments.

Mounted on the center of the frame, the CLOC, fitted with a clear square polycarbonate lens and filled with 75 L of freshwater, (61 cm × 61 cm × 60 cm) was positioned facing horizontally out into the water column at a forward-facing angle between 8° and 10° and fixed onto a back bar on the frame for stability. The weight of the CLOC-BRUV frame and funnel when empty (i.e., not filled with freshwater) was 80 kg. When full, this weight increased to 155 kg. A single Canon high-definition HFG40 camera, fixed with a custom polyvinyl chloride (PVC) housing with clear acrylic view ports was fixed onto the end of the CLOC using a Flexseal 150–165 mm Drainage Coupling DC165. Fins were positioned at the back of the frame for orientating with the prevailing current and a customized rubber gasket, lined with silicone grease, was bolted between the polycarbonate lens and the metal components of the CLOC to keep water tight.

The camera had a 20× HD Video Lens offering a 35 mm equivalent of 26.8–576 mm, resulting in a horizontal field of view of 45.5°. Focal length was set to infinity (∞) which allows for all elements in the field of view to be in focus no matter the distance from the lens. Face detection and tracking, and image stabilization were disabled during deployments (Unsworth et al. 2014). Video data were recorded on to internal Secure Digital (SD) cards. A bait pole was fixed parallel to the CLOC at a distance of 65 cm from the camera and approximately 30 cm from the floor in order for the bag to be comfortably in the field of view of the camera. A 5 mm PVC mesh bait bag was positioned in the center of the field of view, with string attached to pull close to the lens. Past research has shown oily fish to be more effective in attracting mobile fauna (Wraith et al. 2013; Unsworth et al. 2014) and therefore Atlantic Mackerel *Scomber scombrus* was used as bait. Approximately 250 g was used per deployment to eliminate the chance of bait weight limiting the number of taxa attracted to the field of view. Two Anchor light-emitting diode (LED) dive lights (Anchor Dive Lights, www.anchordivelights.com) were mounted above the frame on either side of the bait using cable ties providing white light to illuminate the field of view.

The stereo-BRUV system

The stereo-BRUV system used in this research was the same setup used by Unsworth et al. (2014). A custom-built galvanized steel frame of the dimensions L80 cm × W50 cm × H50 cm and weight of 30 kg was used for these deployments (Fig. 2).

Two Canon high-definition HFG10 cameras fixed within custom PVC housings with clear acrylic viewing ports were positioned on to the frame at an 8° forward facing angle. The separation between the front of the two cameras was 30 cm.

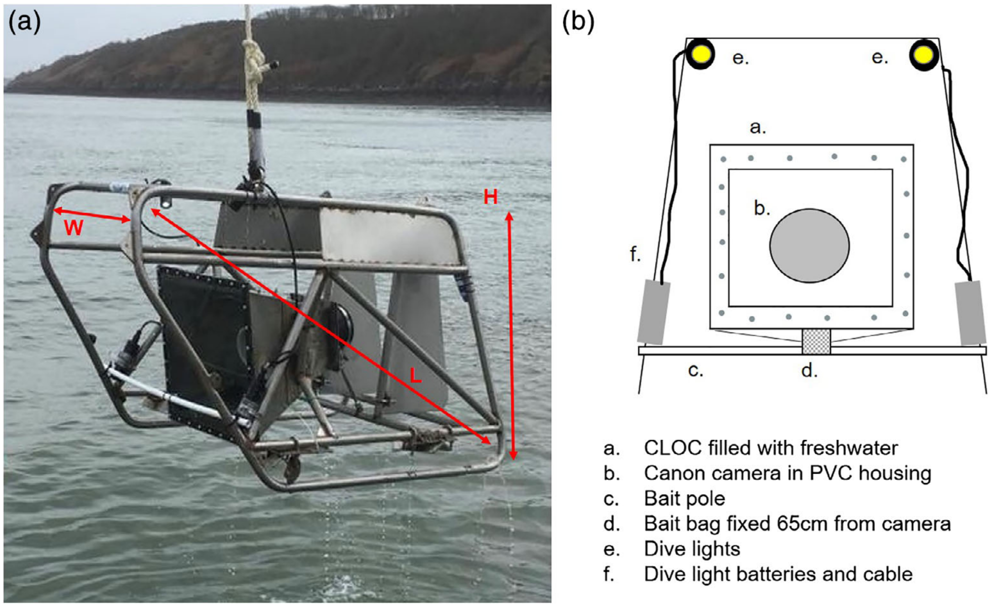

Fig. 1. (**a**) Image of CLOC frame during field deployments. (**b**) Simple schematic of CLOC system setup for field deployments including camera, bait, and light positioning.

Focal length was set to infinity (∞) and face detection and tracking, and image stabilization were disabled during deployments (Unsworth et al. 2014) with a horizontal field of view of 45.5°. Video data were recorded on to internal SD cards. A 65-cm bait pole with a 5-mm mesh bag was mounted in front of the cameras, approximately 15 cm to the floor. Approximately 250 g of *S. scombrus* was also used as bait, illuminated by two Anchor LED dive lights mounted on either side of the frame to provide white light.

The addition of the CLOC to the BRUV system did not impact the horizontal or vertical field of view due to the size of the square polycarbonate lens and field of view specific to the Canon high-definition HFG10 camera. The height of the camera to the seabed in the CLOC-BRUV system was 40 cm compared to 20 cm for the stereo-BRUV system. Although the seabed was still visible when using the CLOC-BRUV system in all deployments, it was considered deeper in the camera field of view.

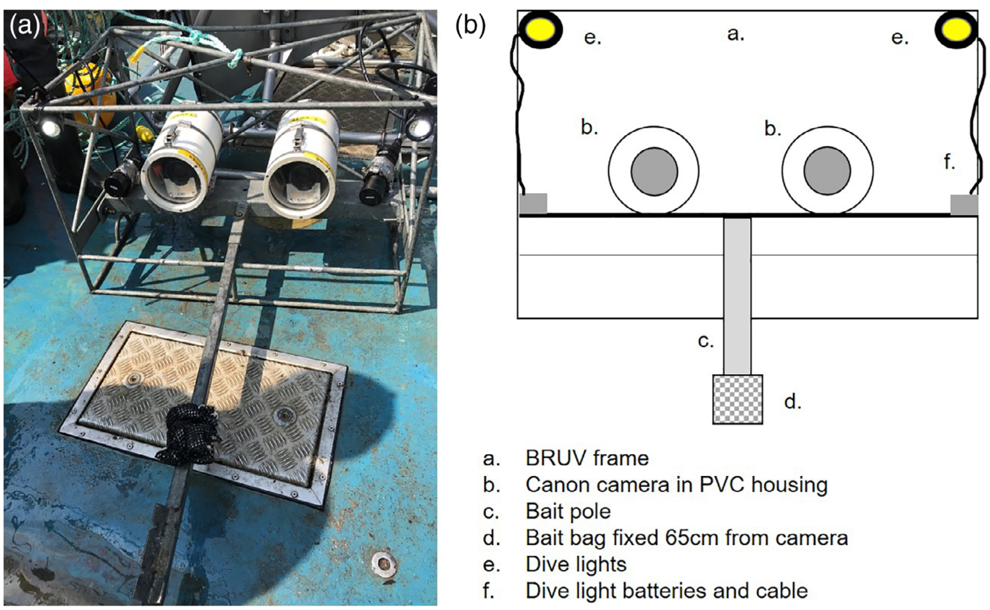

Fig. 2. (**a**) Image of stereo housing system during field deployments. (**b**) Simple schematic of stereo-BRUV system setup for field deployments including camera, bait, and light positioning.

Laboratory trials

Trials comparing the efficiency of a CLOC in a low-visibility environment were carried out under controlled conditions. One cylindrical tank (r = 0.75 m, h = 1.5 m) was filled with approximately 1.78 m^3 of clean freshwater with the two camera systems submerged. Four images of different fish species found in Northern European waters (whiting *Merlangius merlangus*; ballan wrasse *Labrus bergylta*; conger eel *Conger conger*; and lesser spotted dogfish *Scyliorhinus canicula*) were placed 65 cm from the Canon HFG model camera in both systems. Diluted *Chlorella sp.* algae was added to the tank in approximately 5 L batches to reduce water visibility from over 1 m (0 μg^{-1} *Chlorella sp.* per 100 mL) to 0.25 m (3.4 μg^{-1} *Chlorella sp.* per 100 mL). This was calculated by filtering 100 mL of algal water sample taken at the end of the experiment and drying in an oven before measuring the dry weight of the algae. In total, eight different visibility levels were generated between the end points of > 1 m and 0.25 m.

A TMC V2 Power Pump circulating 5400 L h^{-1} was also placed into the tank to keep the algae suspended and mixed into the water column. Due to the shallow nature of the tank, natural light was the only light source present during this experiment. Artificial light was considered, but the glare on the plastic-coated images (for waterproofing) proved excessive for identifying features. Vertical underwater visibility readings were taken at each algal addition using a LaMotte Secchi Disk (www.lamotte.com/en/) with a calibrated line.

Field trials

Comparative BRUV deployments were undertaken at four locations across the United Kingdom (Fig. 3) in areas of varying underwater visibility (Table 1).

For each location, an area of 200 m × 200 m was chosen covering similar depths and substrate types using a combination of skipper's knowledge of the area and existing publicly available benthic habitat maps (European Marine Observation and Data Network 2018). Within this area, the two BRUV systems were randomly deployed simultaneously within a distance of 50 m of each other for a period of 1 h during daylight hours (08:00—18:00) based on previous camera comparison methods (Unsworth et al. 2014; Logan et al. 2017). A distance of 50 m was chosen to reduce the likelihood of motile assemblage numbers and composition differing spatially between simultaneous deployments. To further compensate this, proportions of taxa identified relative to the total number of taxa visits to the camera system during each deployment were calculated for this analysis, as equal abundance visits of taxa to each BRUV system deployments field of view was

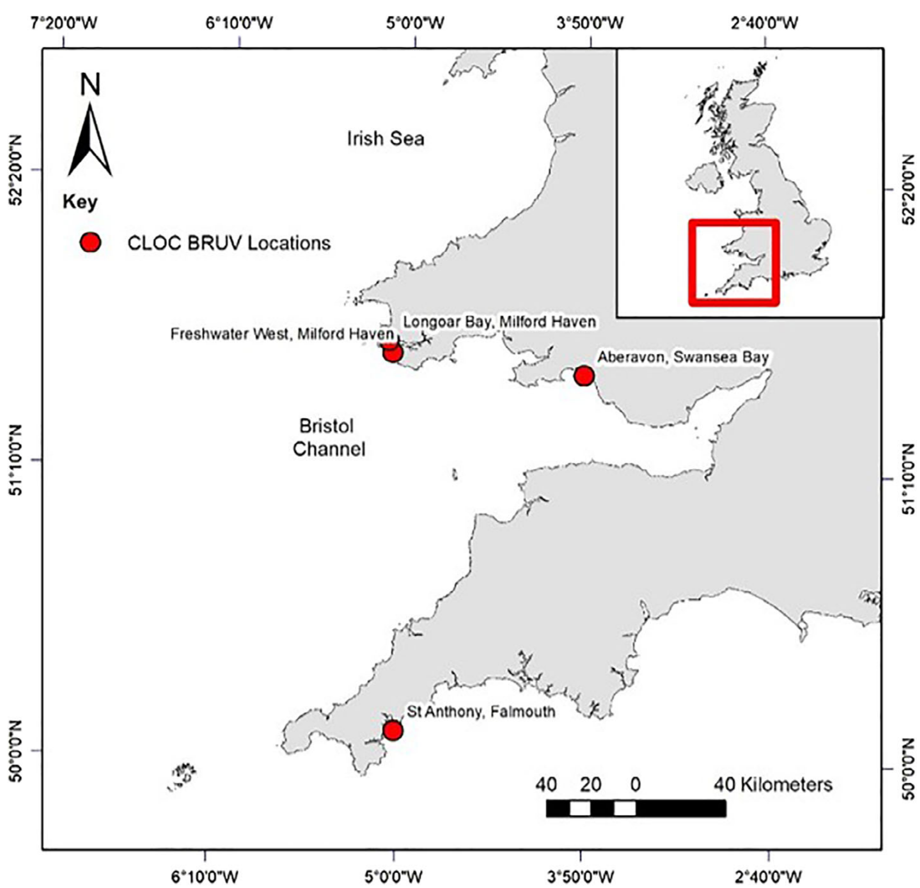

Fig. 3. Map showing the four locations of CLOC and comparison stereo-BRUV deployments taken around the South Wales Coast and South West England (U.K.).

Table 1. Locations and numbers of successful comparative CLOC BRUV and stereo-BRUV deployments taken from the South Wales and South-West England Coasts.

	Latitude	Longitude	CLOC-BRUV deployments	Stereo-BRUV deployments
Longoar Bay, Milford Haven	51°42.761′N	5°06.798′W	3	3
Freshwater West, Milford Haven	51°39.767′N	5°05.265′W	3	3
Aberavon, Swansea Bay	51°35.153′N	3°50.854′W	3	3
St. Anthony, Falmouth	50°08.570′N	5°01.220′W	4	4

Table 2. Number of images for each of the eight visibility levels generated from the addition of 5 L batches of *Chlorella sp.* to the tank.

Visibility level	Stereo-BRUV	CLOC-BRUV
≥ 1 m	6	6
0.85 m	1	1
0.75 m	1	1
0.50 m	2	2
0.40 m	3	3
0.35 m	3	3
0.30 m	5	5
0.25 m	1	1

considered unlikely. A minimum of three deployments of each BRUV system was undertaken at each location. In order to assess underwater visibility, a LaMotte Secchi Disk with a calibrated line was taken for each simultaneous BRUV deployment.

Video analysis

For the laboratory trials, image analysis of the footage was undertaken for both systems at each algal batch at the same time stamp. This totaled 22 images across the eight visibility levels generated (Table 2).

For each of the four fish species at each visibility level, the ability to see prominent identifying features (Yes or No) was assessed using a tailored questionnaire based on identifiable features for each fish species as described in Tyler-Walters (2008) and Henderson (2014).

Analysis of video footage collected in the field followed the same methodology as described by Unsworth et al. (2014) previously used for BRUV work in the United Kingdom. Raw footage was compressed from Advanced Video Coding High Definition format (standard format for digital recordings and high-definition video camcorders) to Audio Video Interleave format using Xilisoft Video/Media Converter Ultimate (www.uk.xilisoft.com). This conversion is required for the use of the footage in the specialist (SeaGIS) software Event Measure (www.seagis.com.au/event.html). This allowed for the footage to be viewed and for the

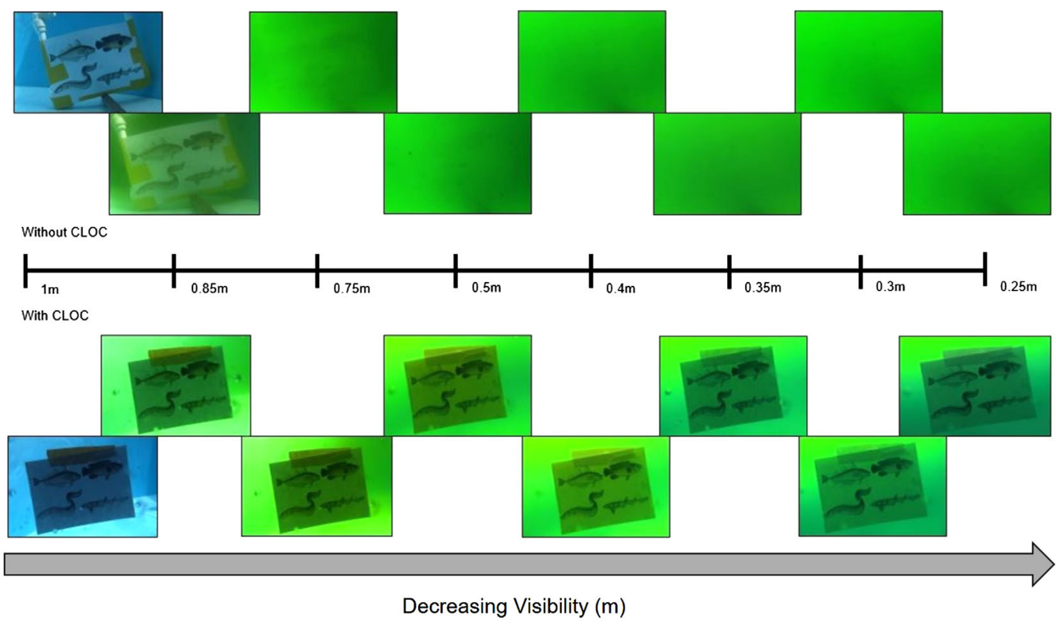

Fig. 4. Observations taken from the recordings of simultaneous deployments of the two BRUV systems across the eight different underwater visibilities measured under controlled conditions using a Secchi disk. Images of the four fish species are positioned 65 cm from the camera in both BRUV system setups.

following analysis: maximum number of individuals observed in one frame (*MaxN*), time of *MaxN*, and arrival time of taxa (Priede et al. 1994; Unsworth et al. 2014) to be conducted.

One analyst with specific experience in both standard video and BRUV data analysis within U.K. coastal waters analyzed the footage from both the laboratory and field experiments to eliminate observer bias between the CLOC and stereo-BRUV data sets. Where taxa could not be confidently identified to species level in the field, a second analyst with additional experience in U.K. faunal assemblages also reviewed to ensure the identification was taken to the highest classification level possible.

Statistical analysis

Summary data are presented as means ± 1 standard error (SE). Statistical analysis was conducted using Minitab 18. Only p values ≤ 0.01 were considered significant to reduce the risk of Type II error due to the small sample sizes.

In order to assess the influence of the CLOC in comparison to a standard BRUV system in controlled conditions, data were Arcsine transformed and a Kruskal–Wallis test was conducted. Both equal variances and normality were not assumed for all four sets of fish feature data collected during this experiment.

A General Linear Model (GLM) was conducted on the Inverse Log transformed field data, following the data passing the Levene test for equal variance, to test the effects of the CLOC on the ability to identify species in varying underwater visibilities. Normality plots of residuals were constructed prior to analysis (Kozak and Piepho 2018). Slight deviations from normality were identified, however, the GLM was considered robust to this (Schmider et al. 2010). A second GLM was conducted on species richness between camera systems with a data passing the Levene test for equal variance and presenting normality. A Kruskal–Wallis test was conducted on arrival times of taxa between camera systems with data in this instance violating the normality assumption.

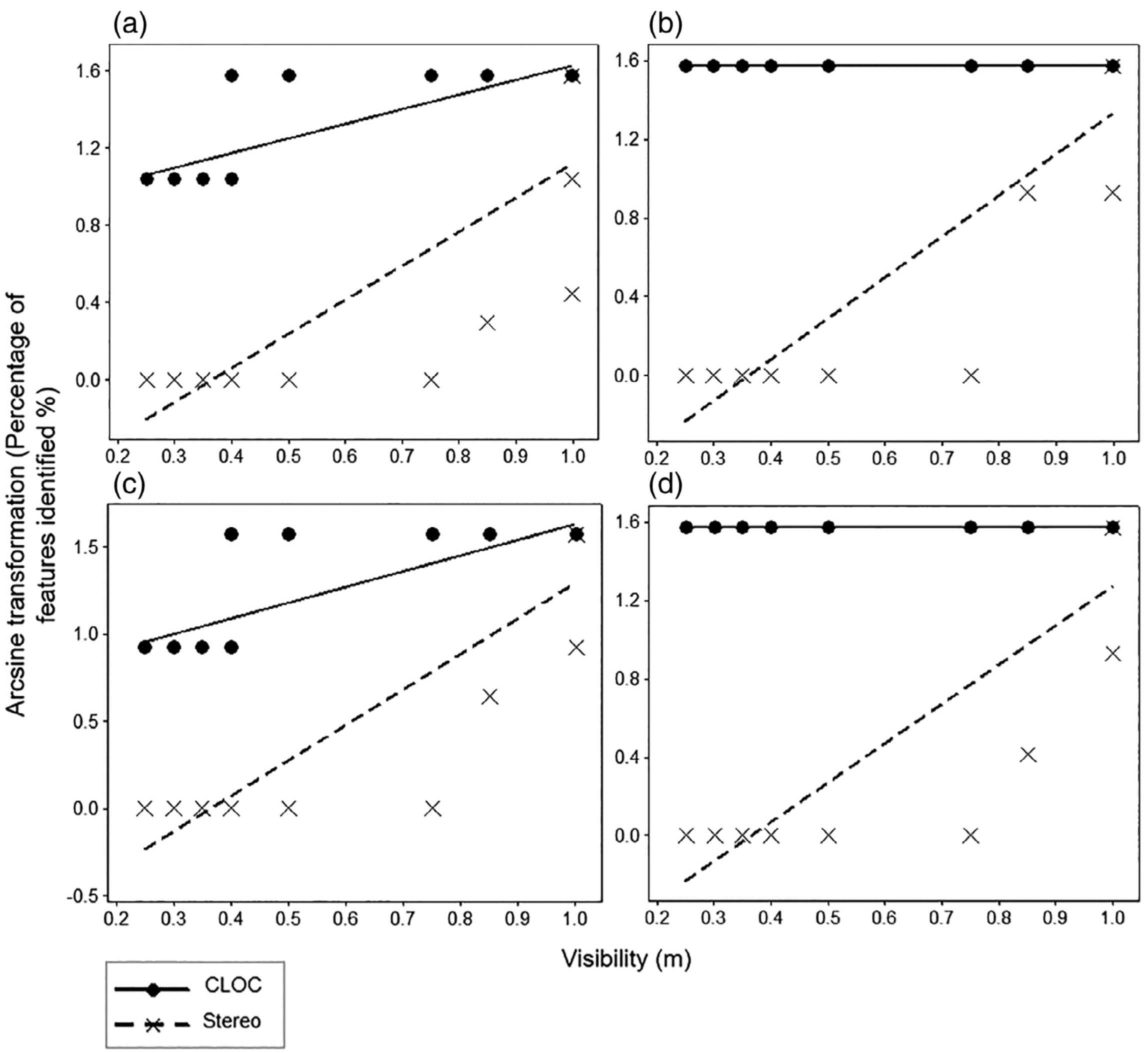

Fig. 5. Percentage of features (after undergoing an Arcsine transformation) successfully identified with a CLOC and without a CLOC for (**a**) whiting, (**b**) ballan wrasse, (**c**) conger eel, and (**d**) lesser spotted dogfish across eight Secchi disk visibility readings (m) in a controlled environment.

Assessment

The following sections illustrate the improvements made to image clarity and the ability to identify taxa to species level in the presence of a CLOC system. First, we address the effectiveness of the CLOC under controlled laboratory conditions for identifying features of fish species. Second, we address the effectiveness of the CLOC in the field for identifying taxa to species level.

Laboratory trials

The controlled trials for this research used a simple approach to prove the concept of the CLOC. Through this approach, underwater image quality and the ability to see the fish images in the presence of a CLOC was greatly increased in comparison to camera deployments without the CLOC in reduced underwater visibility gradients as shown in Fig. 4.

The ability to observe identifying features relating to four different fish species was also improved in the presence of a CLOC in comparison to camera deployments without a CLOC in reduced underwater visibility in all instances (whiting $H_1 = 17.78$, $p \leq 0.001$, ballan wrasse $H_1 = 26.41$, $p \leq 0.001$, conger eel $H_1 = 14.64$, $p \leq 0.001$, and lesser spotted dogfish $H_1 = 26.40$, $p \leq 0.001$) (Fig. 5). In perfect visibility conditions, that is, no algae added into the tank, the ability to identify features without a CLOC was not affected for all four fish species.

Of the four fish species used in this trial, the average ability to identify 100% of features present on the image at all underwater visibility levels with a CLOC occurred for ballan wrasse and lesser spotted dogfish (0.00 ± 1SE). The average ability to identify features for whiting and conger eel with a CLOC was reduced to 90% (0.05 ± 1SE) and 87% (0.07 ± 1SE), respectively, at an underwater visibility of 0.4 m. At an underwater visibility of 0.35 m, the average ability to identify features of whiting further decreased to 86% (0.00 ± 1SE). No further change occurred when visibility was decreased for either whiting or conger eel.

In comparison, the average ability to identify features for all four fish species ranged between 88% and 97% at a visibility of 1 m after algae was added when using the stereo-BRUV system. When visibility was reduced to 0.85 cm, the ability to identify features associated with whiting, ballan wrasse, conger eel, and lesser spotted dogfish reduced to 29%, 80%, 60%, and 40%, respectively (0.00 ± 1SE). At a visibility of 0.75 cm and below, the average ability to identify features for all four fish species was zero.

Field trials

The two BRUV systems in the field presented similar results to those identified under controlled conditions. Underwater image clarity again showed an improvement at low-visibility levels, with the ability to see and identify taxa at a distance of 65 cm enhanced when using the CLOC (Fig. 6). For instance, at an underwater visibility of 3.5 m, a lesser spotted dogfish was clearly visible when utilizing a CLOC-BRUV system. An individual was seen using the stereo-BRUV system during the same deployment although it was much more difficult to see any identifying features. The size and weight of the CLOC-BRUV system had little influence on the resuspension of sediments into the water column upon deployment on the seabed. The sediment settling time for the stereo-BRUV system across the 13 field deployments was 23.66 ± 11.2 s compared to 22.8 ± 4.2 s for the CLOC-BRUV system.

Comparisons of the proportions of taxa successfully identified using a CLOC-BRUV system to the stereo-BRUV system across a gradient of visibilities are presented in Fig. 7 with the

Fig. 6. Observations of a taxa taken from comparative simultaneous deployments without the CLOC (left) and with the CLOC (right) where a lesser spotted dogfish is visible in a location of low underwater visibility (3.5 m) in Swansea Bay, South Wales. An individual is also present in the imagery without the CLOC; however, distinguishing features are difficult to determine.

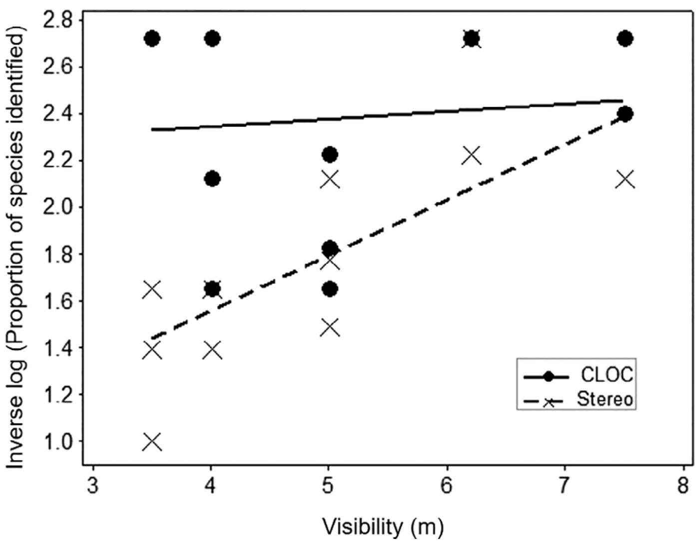

Fig. 7. Proportions of species (after undergoing and inverse log transformation) identified from the taxa recorded during comparative camera system deployments with and without a CLOC across four locations of varying underwater visibility (m) taken from the South Wales and South-West England coastlines (U.K.).

introduction of a CLOC-BRUV system significantly influencing the ability to identify taxa to species level ($F_{1,24} = 11.25$, $p \leq 0.01$). Fewer differences were seen between the proportions at higher underwater visibilities (5 m and above) for the two BRUV systems. However, when underwater visibility was reduced to 4 m and below, differences in proportions of taxa identified to species level were apparent.

Comparisons between the species richness and arrival times of taxa into the camera field of view for both camera systems is shown in Fig. 8. No statistical differences were identified between the two systems for both species richness ($F_{1,24} = 0.20$, $p = 0.66$) and arrival times of taxa ($H_1 = 1.64$, $p = 0.20$). The size and shape of the CLOC-BRUV system is therefore not expected to lead to significant differences in taxa numbers or arrival times but only increase the taxonomic level in which it is identified to. It may also be deployed for the same duration of 1 h as standard BRUV systems.

Discussion

The introduction of a CLOC to a conventional BRUV system in low-visibility environments improved depth of vision and image clarity as well as increased species level identification relative to traditional BRUV systems.

Feasibility

Underwater cameras are a common and nondestructive tool for environmental assessments. The introduction of a CLOC to more conventional BRUV camera systems may be considered a simple and reliable way of broadening the operational window for underwater cameras in turbid environments. During this study, any failed deployments of the CLOC-BRUV system were due to human error with regards to camera settings, and not the CLOC-BRUV system itself. For this research, Canon model cameras were used for consistency with our existing stereo system

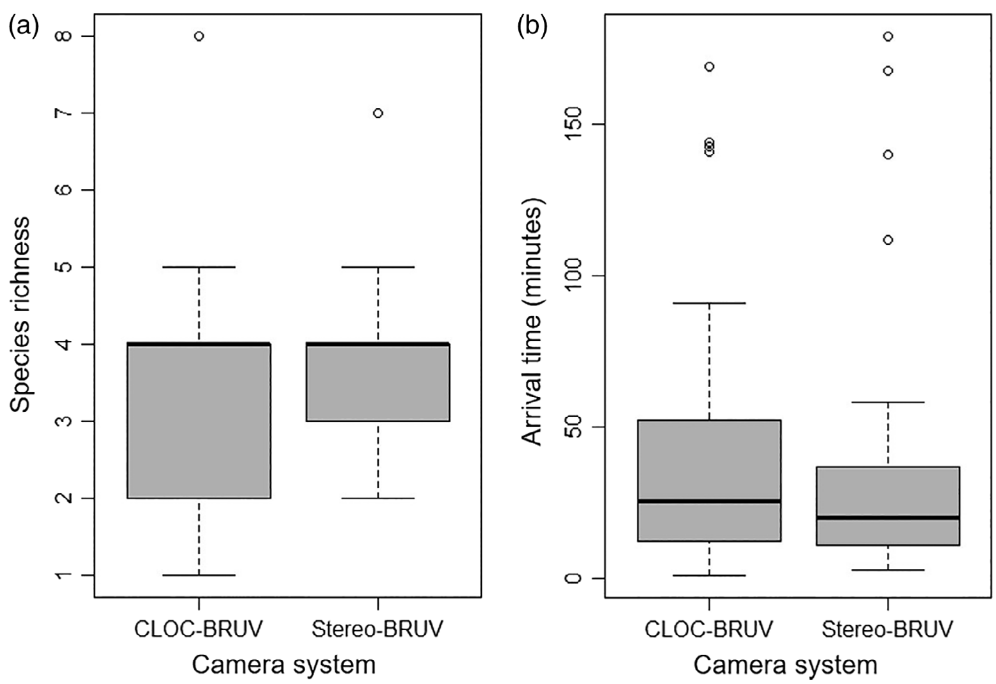

Fig. 8. Boxplot (box ranging from first to third quartile and highlighting median value, whiskers extending to 1.5 the interquartile distance with circles indicating outliers) showing the (**a**) species richness* and (**b**) arrival times of taxa into the camera frame for the two camera systems stereo-BRUV and CLOC-BRUV. *Species richness refers to all taxa recorded and identified to the highest taxonomic level possible.

and resources; however, the design of the CLOC is customizable and could be amended for use with other camera housing sizes such as those required for the use of compact action cameras or similar camera types. However, the wider field of view of these compact cameras must be taken into consideration in relation to the funnel housing. The CLOC-BRUV system used for this research had been custom made to fit with a number of different camera manufacturers including Kongsberg and Rovtech as it was in use by several organizations for varying needs. Due to the size and weight of the CLOC-BRUV frame when filled with freshwater, it was not possible to hand haul the system to the seafloor as commonly practiced when using lighter mono-BRUV systems (Esteban et al. 2018; Jones et al. 2018). Vessels equipped with A frames and hydraulic winches were therefore required costing significantly more than smaller craft used when hand hauling. The cost of employing such vessels is therefore an important factor to consider when proposing future research using CLOC-BRUV systems.

The materials used to build the frame for the CLOC-BRUV system are considered to be highly durable for use in a number of harsh environments both freshwater and marine. Maintenance of this system is minimal, with only a wash down with freshwater needed post deployment and replacement of materials such as the polycarbonate lens as and when required. Consumables such as cable ties, bait bag mesh, and string used to attach the bait and dive lights to the frame are considered low in cost and easy to replace.

Alternative camera methods

Other methods of visualizing motile fauna in low-underwater visibilities include sonar technology. Sonar cameras are increasingly being used in turbid conditions for structural assessments in the oil and gas industry as well as monitoring of known migratory fish assemblages in rivers. The acquisition of this technology is expensive in comparison to the CLOC-BRUV system and may also be susceptible to acoustic backscatter in areas of high turbulence especially in coastal areas with large tidal ranges, strong tidal flows, and sediment-laden waters (Melvin and Cochrane 2014). These methods, although offering considerable potential for biodiversity assessments remain unproven in their capacity to accurately identify fish species in the field (Martignac et al. 2015).

Acoustic survey methods alone are not currently an adequate method of accurately assessing biodiversity; ground-truthing through the use of camera footage using a CLOC system may therefore be useful addition when assessing biological community composition and specific species abundance (McClatchie et al. 2000; Mackinson et al. 2002; Brown et al. 2011; Martignac et al. 2015).

Practical application

CLOC-BRUV systems may be applied in both riverine and marine waters globally, targeting fish assemblages and/or motile benthic macrofauna. Likely applications of this system would be assessing community composition, distribution, and relative abundance of motile fauna, particularly in poorly studied habitat that are commonly highly turbid including mangrove areas where BRUV methods have previously been implemented (Benzeev et al. 2017; Enchelmaier et al. 2018) and salt marshes. Furthermore, due to its customizable design, this CLOC-BRUV system may also be used in the application of benthic habitat assessment in turbid environments as a drop-down camera mirroring those already in use by the marine surveying industry (Hitchin et al. 2015).

The remote deployment of the CLOC also provides health and safety benefits as current close-range visual surveys of motile fauna in low-visibility environments may involve the use of divers. Such diver surveys are depth and time restricted, and diving in these turbid conditions may be considered dangerous with potential hazards including underwater currents, tides, pollution, and dangerous aquatic fauna putting the diver at risk. A CLOC-BRUV system provides a safer and remote alternative to this.

As the CLOC-BRUV system aims to improve the image clarity in the immediate vicinity of the bait, it is advised that this system is used in underwater visibility levels of 4 m and below when measuring water visibility using a Secchi Disk. Differences in underwater visibility at the surface of the water column and at the seabed must be taken into consideration when using this method with visibility at the seabed usually lower with factors such as sea state, surface glare, cloud cover, and human bias influencing readings (Davies-Colley 1988). In order to achieve a more accurate assessment of underwater visibility at the seabed, a turbidity profiler or total suspended solid sampling should be used. Methods used in the controlled assessment during this research tested the influence of visibility based on levels of diluted *Chlorella sp*. algae. This is thought to be representative of the restrictive visibility created by plankton blooms when deploying underwater cameras. Under the same scenario of suspended sediments and/or organic matter influencing underwater visibility, the same results are expected through the presence of a CLOC-BRUV system over traditional BRUV methods based on reviews of field video footage. The suspension of sediments into the water column as the frame landed on the seabed had little or no impact on the clarity of the image and lasted on average for 22.8 ± 4.2 s.

With the value of gaining length and biomass estimates through the increased use of stereo-BRUV applications high, the potential of using a CLOC in stereo BRUV systems have previously been discussed during this research. A larger chamber positioned in a rectangular shape across both cameras would be required for this which in turn would require more freshwater and therefore add more weight to the system. Calibrating such a system would also require a large vessel or lifting gear and divers for deployments as the chamber would need to be filled with freshwater during this process and calibrated in either the field or a freshwater pool large enough to accommodate both the CLOC and calibration equipment. If undertaking calibrations in the field, adequate visibility levels would also need to be required in order to see the calibration cube or other structure such as a distance bar used during the calibration process. Calibrations in a freshwater pool would increase calibration stability due to the lack of currents, tides, and weather influences.

Conservation relevance

The use of a CLOC-BRUV system has the ability to improve the conservation and management of fauna associated with sensitive habitats in protected areas. A quantitative measure of biodiversity through a diversity index is essential when presenting information through an environmental baseline survey of an area. These indices may include species richness (R), Shannon Index of Diversity (H^1), Simpson's Index of Diversity (λ), and Species Evenness (J') (Gray 2000). As species are usually the interest when trying to characterize an area, acquiring good quality footage allowing for the successful identification of taxa to the species level is required while minimizing disturbance and/or damage to the target species or habitat.

This may be crucial in the decision-making process regarding the conservation objectives of designated protected sites, creating population targets, and understanding the natural variation in community composition and population.

With increases in coastal developments globally, there is also a need to implement a simple, reliable, safe, and repeatable monitoring method for faunal communities associated with these developments. A CLOC-BRUV system allows for this in turbid and highly dynamic environments, with static deployments minimizing risk of damage to existing seabed infrastructure.

Overall, this research has been successful in proving the concept of a CLOC-BRUV system and further moving forward the applicability of underwater cameras in low-visibility aquatic environments.

Data availability statement

Raw data will be uploaded to https://pangaea.de/

References

Benzeev, R., N. Hutchinson, and D. A. Friess. 2017. Quantifying fisheries ecosystem services of mangroves and tropical artificial urban shorelines. Hydrobiologia **803**: 225–237. doi:10.1007/s10750-017-3299-8

Bicknell, A. W., B. J. Godley, E. V. Sheehan, S. C. Votier, and M. J. Witt. 2016. Camera technology for monitoring marine biodiversity and human impact. Front. Ecol. Environ. **14**: 424–432. doi:10.1002/fee.1322

Brown, C. J., S. J. Smith, P. Lawton, and J. T. Anderson. 2011. Benthic habitat mapping: A review of progress towards improved understanding of the spatial ecology of the seafloor using acoustic techniques. Estuar. Coast. Shelf Sci. **92**: 502–520. doi:10.1016/J.ECSS.2011.02.007

Cappo, M., E. S. Harvey, and M. Shortis. 2006. Counting and measuring fish with baited video techniques - An overview. AFSB Conference and Workshop, Cutting-Edge Technologies in Fish and Fisheries Science. **1**: 101–114.

Cappo, M., P. Speare, and G. De'Ath. 2004. Comparison of baited remote underwater video stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the Great Barrier Reef Marine Park. J. Exp. Mar. Biol. Ecol. **302**: 123–152. doi:10.1016/j.jembe.2003.10.006

Davies-Colley, R. J. 1988. Measuring water clarity with a black disk. Limnol. Oceanogr. **33**: 616–623. doi:10.4319/lo.1988.33.4.0616

Davies, J., and others. 2001. Marine monitoring handbook. Joint Nat. Conserv. Comm. **11**: 405–487. doi:10.1016/0141-9331(87)90207-9

Veritas D. N. 2010. DNV-RP-F111: Interference between trawl gear and pipelines. 1–35.

Enchelmaier, A. C., E. A. Babcock, and N. Hammerschlag. 2018. Survey of fishes within a restored mangrove habitat of a subtropical bay. Estuar. Coast. Shelf Sci. (in press). doi:10.1016/J.ECSS.2018.11.009

Esteban, N., R. K. F. Unsworth, J. B. Q. Gourlay, and G. C. Hays. 2018. The discovery of deep-water seagrass meadows in a pristine Indian Ocean wilderness revealed by tracking green turtles. Mar. Pollut. Bull. **134**: 99–105. doi:10.1016/J.MARPOLBUL.2018.03.018

European Marine Observation and Data Network. 2018. Seabed habitats interactive map.

Evans, P. G. H., and L. Thomas. 2011. Estimation of costs associated with implementing dedicated cetacean surveillance scheme in UK.

Gill, A. B. 2005. Offshore renewable energy: Ecological implications of generating electricity in the coastal zone. J. Appl. Ecol. **42**: 605–615. doi:10.1111/j.1365-2664.2005.01060.x

Gray, J. S. 2000. The measurement of marine species diversity, with an application to the benthic fauna of the Norwegian continental shelf. J. Exp. Mar. Biol. Ecol. **250**: 23–49. doi:10.1016/S0022-0981(00)00178-7

Griffin, R. A., G. J. Robinson, A. West, I. T. Gloyne-Phillips, and R. K. F. Unsworth. 2016. Assessing fish and motile fauna around offshore windfarms using stereo baited video V. PLoS One **11**: e0149701. doi:10.1371/journal.pone.0149701

Heiskanen, A.-S., and others. 2016. Biodiversity in marine ecosystems—European developments toward robust assessments. Front. Mar. Sci. **3**: 184. doi:10.3389/fmars.2016.00184

Henderson, P. 2014. Identification guide to inshore fish of the British isles. Pisces Conservation.

Hitchin, R., J. A. Turner, and E. Verling. 2015. Epibiota remote monitoring from digital imagery: Operational guidelines.

Inger, R., and others. 2009. Marine renewable energy: Potential benefits to biodiversity? An urgent call for research. J. Appl. Ecol. **46**: 1145–1153. doi:10.1111/j.1365-2664.2009.01697.x

Jones, B. L., L. C. Cullen-Unsworth, R. Howard, and R. K. F. Unsworth. 2018. Complex yet fauna-deficient seagrass ecosystems at risk in southern Myanmar. Bot. Mar. **61**: 193–203. doi:10.1515/bot-2017-0082

Kozak, M., and H. P. Piepho. 2018. What's normal anyway? Residual plots are more telling than significance tests when checking ANOVA assumptions. J. Agron. Crop Sci. **204**: 86–98. doi:10.1111/jac.12220

Levin, L. A., and others. 2001. The function of marine critical transition zones and the importance of sediment biodiversity. Ecosystems **4**: 430–451. doi:10.1007/s10021-001-0021-4

Lindholm, J., M. Gleason, D. Kline, L. Clary, S. Rienecke, A. Cramer, and M. Los Huertos. 2015. Ecological effects of bottom trawling on the structural attributes of fish habitat in unconsolidated sediments along the central California outer continental shelf. Fish. Bull. **113**: 82–96. doi:10.7755/FB.113.1.8

Logan, J. M., M. A. Young, E. S. Harvey, A. C. G. Schimel, and D. Ierodiaconou. 2017. Combining underwater video methods improves effectiveness of demersal fish assemblage surveys across habitats. Mar. Ecol. Prog. Ser. **582**: 181–200. doi:10.3354/meps12326

Mackinson, S., S. Freeman, R. Flatt, and B. Meadows. 2002. Improving acoustic surveys by combining fisheries and ground discrimination acoustics. ICES Doc. C.: 1–9.

Mallet, D., and D. Pelletier. 2014. Underwater video techniques for observing coastal marine biodiversity: A review of sixty years of publications (1952–2012). Fish. Res. **154**: 44–62. doi:10.1016/j.fishres.2014.01.019

Martignac, F., A. Daroux, J. L. Bagliniere, D. Ombredane, and J. Guillard. 2015. The use of acoustic cameras in shallow waters: New hydroacoustic tools for monitoring migratory fish population. A review of DIDSON technology. Fish Fish. **16**: 486–510. doi:10.1111/faf.12071

McClatchie, S., R. E. Thorne, P. Grimes, and S. Hanchet. 2000. Ground truth and target identification for fisheries acoustics. Fish. Res. **47**: 173–191. doi:10.1016/S0165-7836(00)00168-5

Mélin, F., and V. Vantrepotte. 2015. How optically diverse is the coastal ocean? Remote Sens. Environ. **160**: 235–251. doi:10.1016/j.rse.2015.01.023

Melvin, G. D., and N. A. Cochrane. 2014. Multibeam acoustic detection of fish and water column targets at high-flow sites. Estuaries Coasts **38**: 227–240. doi:10.1007/s12237-014-9828-z

Meybeck, M. 1993. Riverine transport of atmospheric carbon: Sources, global typology and budget. Water Air Soil Pollut. **70**: 443–463. doi:10.1007/BF01105015

Pelc, R., and R. M. Fujita. 2002. Renewable energy from the ocean. Mar. Policy **26**: 471–479. doi:10.1016/S0308-597X(02)00045-3

Priede, I. G., P. M. Ragley, and K. L. Smith. 1994. Seasonal change in activity of abyssal demersal scavenging grenadiers Coryphaenoides (Nematonums) armatus in the eastern North Pacific Ocean. Limnol. Oceanogr. **39**: 279–285. doi:10.4319/lo.1994.39.2.0279

Schmider, E., M. Ziegler, E. Danay, L. Beyer, and M. Bühner. 2010. Is it really robust?: Reinvestigating the robustness of ANOVA against violations of the normal distribution assumption. Methodology **6**: 147–151. doi:10.1027/1614-2241/a000016

Sheehan, E. V., T. F. Stevens, and M. J. Attrill. 2010. A quantitative, non-destructive methodology for habitat characterisation and benthic monitoring at offshore renewable energy developments. PLoS One **5**: e14461. doi:10.1371/journal.pone.0014461

Smith, V. H. 2003. Eutrophication of freshwater and coastal marine ecosystems: A global problem. Environ. Sci. Pollut. Res. **10**: 126–139. doi:10.1065/espr2002.12.142

Tyler-Walters, H. 2008. MarLIN - The marine life information network. Biology and sensitivity key information subprogramme [Internet].

Unsworth, R. K. F., J. R. Peters, R. M. McCloskey, and S. L. Hinder. 2014. Optimising stereo baited underwater video for sampling fish and invertebrates in temperate coastal habitats. Estuar. Coast. Shelf Sci. **150**: 281–287. doi:10.1016/j.ecss.2014.03.020

Watson, D. L., E. S. Harvey, M. J. Anderson, and G. A. Kendrick. 2005. A comparison of temperate reef fish assemblages recorded by three underwater stereo-video techniques. Mar. Biol. **148**: 415–425. doi:10.1007/s00227-005-0090-6

Whitmarsh, S. K., P. G. Fairweather, and C. Huveneers. 2017. What is big BRUVver up to? Methods and uses of baited underwater video. Rev. Fish Biol. Fish. **27**: 53–73. doi:10.1007/s11160-016-9450-1

Wilber, D. H., and D. G. Clarke. 2001. Biological effects of suspended sediments: A review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. N. Am. J. Fish. Manag. **21**: 855–875. doi:10.1577/1548-8675(2001)0212.0.CO

Wraith, J., T. Lynch, T. E. Minchinton, A. Broad, and A. R. Davis. 2013. Bait type affects fish assemblages and feeding guilds observed at baited remote underwater video stations. Mar. Ecol. Prog. Ser. **477**: 189–199. doi:10.3354/meps10137

Acknowledgments

We would like to acknowledge the support and funding given by Knowledge Economy Skills Scholarships (KESS). This is a pan-Wales higher level skills initiative led by Bangor University on behalf of the HE sector in Wales. It is partly funded by the Welsh Government's European Social Fund (ESF) convergence program for West Wales and the Valleys. We would also like to acknowledge the support and funding given by Ocean Ecology Ltd and EU EDRF funding through the European Union Regional Development Fund to the project SEACAMS2. Further thanks go to Keith Naylor, Max Robinson, Nick Fleming, Bettina Walter, and Robert Smith (Swansea University), Haven Marine Surveying, and Trevor Butler (Penderyn Skipper).

Conflict of Interest

None declared.

Submitted 11 February 2019
Revised 22 July 2019
Accepted 09 August 2019

Associate editor: George Waldbusser